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Ecological Assessment of the Beneficial Use of Dredged Sediments for Island Restoration: Mordecai Island, Barnegat Bay, NJ

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Ecological Assessment of the Beneficial Use of Dredged Sediments for Island Restoration: Mordecai Island, Barnegat Bay, NJ

October, 2020

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1. Introduction

Navigation channels across the country require regular maintenance dredging to avoid infilling with sediments and becoming impassible by nautical vessels. In an experimental strategy that is often referred to as “beneficial use,” sediments removed during maintenance dredging operations can be used to support the creation and enhancement of local coastal habitats. Vegetated coastal habitats can be effective in stabilizing shorelines against erosion and can help to protect against storm-induced flooding (Narayan et al. 2017). These same habitats also provide a wide range of ecosystem service benefits (e.g., water quality remediation, carbon storage, habitat for wildlife). As a result, it is desirable to use natural or created habitats, including those created using dredge materials in place of traditional engineered structures for coastal protection where possible (Temmerman et al. 2013, Sutton-Grier et al. 2015). When natural coastal features like islands are used in place of engineered structures like seawalls for coastal protection, they are often referred to as Natural Infrastructure or Natural and Nature Based Features (NNBF); terms used to emphasize the role these habitats play in protecting developed shorelines from coastal hazards. The goals of beneficial use projects include low-cost sediment disposal, enhancement of ecological function, and maximizing the role that natural coastal habitats play in mitigating storm surge and flood risk and stabilizing shorelines against erosion. To date, the protective and ecosystem service benefits of such projects have not been well-quantified and the lack of data on long-term performance has been cited as a barrier to their widespread implementation (Spalding et al. 2014, Arkema et al. 2015).

In partnership with the U.S. Army Corps of Engineers (USACE) Engineering With Nature (EWN®) initiative, the Philadelphia District of USACE is committed to serving as a proving ground for the use of natural and nature-based approaches to coastal protection. Through this role, in 2015, USACE Philadelphia used sediment dredged from the New Jersey Intracoastal Waterway to restore Mordecai Island in a demonstration of the beneficial use of sediments to restore intertidal habitat and enhance the protective value of the island to adjacent developed shorelines. Mordecai is an undeveloped island that runs parallel to the Barnegat Bay shoreline of Beach Haven, New Jersey. In addition to providing habitat for a wide range of estuarine organisms and nesting shorebirds (Burger et al. 2001), the island serves as a wave-break, protecting the adjacent developed shoreline of Beach Haven from the erosive action of waves generated in Barnegat Bay. Over the past century, persistent wave action has taken a toll on Mordecai Island, resulting in erosion of the western shoreline and leading to a breach that effectively separated the island into two lobes. Without action, continued erosion-induced widening and deepening of the breach would have led to increased wave-exposure for the mainland shoreline.

The restoration involved the beneficial use of approximately 30,000 cubic yards of dredged sediment to reconnect the two lobes of the island. The final design of the placement area included a central high-elevation mound intended to provide habitat for shorebird nesting. The region surrounding the central mound was filled to intertidal elevations to create a seamless transition between the placement area and the two marsh-dominated lobes of Mordecai Island that it joined (Figure 1). The placement area was planted with native salt marsh and transitional/upland vegetation. Goose exclusion fencing (a grid of wooden stakes connected with monofilament line) was placed throughout the planted region to discourage herbivory. In addition to sediment placement activities, local partners including the Mordecai

Land Trust have installed a variety of wave-attenuating devices along the island's western shoreline in an attempt to protect against further erosive losses. These devices include large, tubular sand-filled bags (aka "geotubes") that parallel the southwestern-most shoreline, and a series of oyster castle structures built with varying configurations of concrete blocks that parallel the shore to the north of the geotubes.

Figure 1. Aerial images depicting Mordecai Island before (a) and after (b) dredged sediment placement. Image source: ESRI World Imagery; 02-20-2014 and 01-08-2018.

Scientists from the NOAA National Centers for Coastal Ocean Science (NCCOS) documented sediment and soil characteristics, erosion rates and biological communities within the sediment placement region and in undisturbed inter- and subtidal regions of Mordecai Island from 2017 – 2019. The goal of these efforts was to quantify the immediate success of restoration efforts, evaluate ecological benefits and impacts of the project, and provide a baseline against which to measure future changes. This report summarizes the results of those efforts and provides a snapshot of the morphological and ecological status of Mordecai Island during that time. The work was funded through discretionary science funding from NOAA/NCCOS with supplemental support in the form of laboratory analysis provided by the USACE Engineering Research and Development Center.

1.1. Monitoring Goals

At the most basic level, success of a dredged material placement project is measured by the degree to which the created feature retains its designed profile over time. The key to stabilization of placed sediments in the intertidal zone, without the use of physical barriers such as containment dikes, is colonization with vegetation. Aboveground stems and leaves of intertidal vegetation function to dampen incoming wave energy while the roots and rhizomes physically bind sediments in place (Coops et al 1996, Leonard and Croft 2006). As a result, the quicker and more completely the placed sediments become colonized with vegetation, the more stable the feature is likely to be. In vegetated wetlands, plants are the building blocks of the three-dimensional habitat structure; their presence and abundance is linked to habitat function. Thus documenting the pace and extent of vegetative colonization within the placement region and stability of the placed sediments was a priority. In addition, we sought to characterize the ecological benefits and impacts of the Mordecai Island restoration to contribute to a broader understanding of the efficacy of such projects. We approached this task by asking three specific questions:

1. **Is the created marsh on a trajectory to develop characteristics of the natural marsh?**

   The ultimate goal of ecosystem restoration is to create habitat that is indistinguishable from nearby reference habitats of the same type. A created site that is indistinguishable from reference habitats can reasonably be assumed to be providing similar ecosystem services and at the same approximate rate. While many features of created marshes may take decades to reach full equivalency with their natural counterparts (Craft et al. 1999, Craft et al. 2003), others, like vegetative cover, often develop rapidly (Currin et al. 2008). As vegetation colonizes newly placed sediments, production and decay of plant material changes the physical character of the placed sediments and influences the biogeochemical processes occurring within the sediments and interstitial waters (Davis et al. 2015). Comparisons of vegetative community and soil structure between created and natural regions of Mordecai (those that were not impacted by the dredge material placement activities) were undertaken to evaluate the pace and trajectory of development of the newly created intertidal habitat.

2. **Did sediment placement result in measurable impacts to the adjacent subtidal benthic infaunal community?**

   Estuarine habitats are subjected to frequent wave, wind, and current-induced disturbances (Wilber and Clarke 2007) and are characterized by low environmental constancy and predictability (Boesch 1974). Estuarine benthic communities in less predictable environments generally have a high proportion of "r-strategists" with high reproductive potential, and are thus better adapted to recover quickly from local disturbances (Boesch 1974, Dauer 1984). Although dredged material placement activities at Mordecai Island were focused on creating intertidal, vegetated marsh habitat, some of the applied sediments could be washed from the placement area onto adjacent subtidal benthic habitats both during the initial placement activities and as a result of erosion of placed sediments, despite efforts to contain the applied material through the use of sediment curtains and clam bags. Prior studies of beneficial use of dredge material to construct or enhance marsh habitat have shown, however, that some benthic organisms (burrowing polychaetes, amphipods, and molluscs) can migrate vertically through dredged material depths of up to 15 - 30 cm (Maurer et al. 1981, 1982; Miller et al. 2002; Roberts et al. 1998; Wilber and Clarke 2007). When the spatial scale of disturbance is limited, adult immigration from surrounding unaffected areas can also contribute to a rapid rate of recovery (Dauer 1984). A benthic sampling component was incorporated into Mordecai Island monitoring activities to evaluate whether any measurable impacts of sediment placement activities on subtidal infaunal communities could be detected.

3. Have the shoreline stabilization and sediment placement activities at Mordecai helped to fortify the Island against further losses?

Anecdotal evidence and historical imagery suggest that the western shoreline of Mordecai Island has retreated over the past century, resulting in a substantial decrease in total areal extent of the Island. The breach represented an additional historic loss in areal extent of Mordecai Island and left the narrow northern tip of the island isolated from the larger southern portion, and both portions exposed to wave erosion on all sides. The overarching goal of the sediment placement action at Mordecai was to stabilize the island against further losses so that it will continue to serve as a physical barrier, protecting the western shoreline of Beach Haven from wave energy. Measurement of changes in shoreline position and total areal extent of Mordecai Island over time will be necessary to evaluate the degree to which these management activities have been effective at slowing the rate of loss.

The answers to these questions are pivotal not only for understanding the implications of restoration activities for the future of Mordecai Island, but for informing the siting and design of dredge-material based restoration projects nationwide.

2. Methods

Field sampling involved a combination of point-based sampling of intertidal and nearshore subtidal regions and targeted surveys of specific regions of interest. Sampling was conducted October 10 – 14, 2017; October 1 – 6, 2018; and September 9 – 12, 2019. The initial sediment placement (approximately 30,000 cubic yards) occurred in late December of 2015. In the spring of 2016 the placement area was planted with native vegetation in a grid pattern with 1 ft centers to stabilize the placed sediments and boost the pace of re-vegetation. An additional placement of approximately 1000 cubic yards was conducted in December of 2017 to elevate the top of the central mound, which was intended to provide nesting habitat for migratory birds, in response to multiple high-tide events that flooded the entire placement region and led to nest failure in the summer and fall of 2017. This secondary placement impacted an isolated area within the original placement region which was at elevations > 2 ft Mean Sea Level (MSL) prior to the second round of dredge material placement and, therefore, above the elevation range of our point-based sampling. While this additional sediment placement ultimately changed the elevation profile of the mound, it had minimal impact on the overall course of development of the wetland community within the larger placement region.

2.1. Point-based Sampling

We used a stratified random sampling design that involved dividing the island and surrounding subtidal regions into two lateral lobes (resulting in east and west halves) and then into three sections lengthwise (North, Central and South). Each of the 6 resulting regions (NW, NE, CW, CE, SW, SE) were divided into two strata by elevation (0 – 1 ft and 1 – 2 ft MSL) for the marsh sampling or depth (0 – 3 ft and 3 – 6 ft water depth) for subtidal sampling, and sample points within each stratum were selected randomly (Figure 2). Strata determinations were based on analysis of LiDAR and bathymetry files available from the U.S. Geological Survey Coastal National Elevation Dataset (CoNED)-Topobathymetric Digital Elevation Model (TDEM) and thus subject to the error associated with these datasets. Measurement of actual sample depths or marsh elevations at each point revealed that they were usually, but not always within the intended depth or elevation range. In 2017 and 2018 a total of 72 intertidal sampling points (12 per region) and 60 subtidal sampling points (12 per region, west; 8 per region, east) were sampled each year.

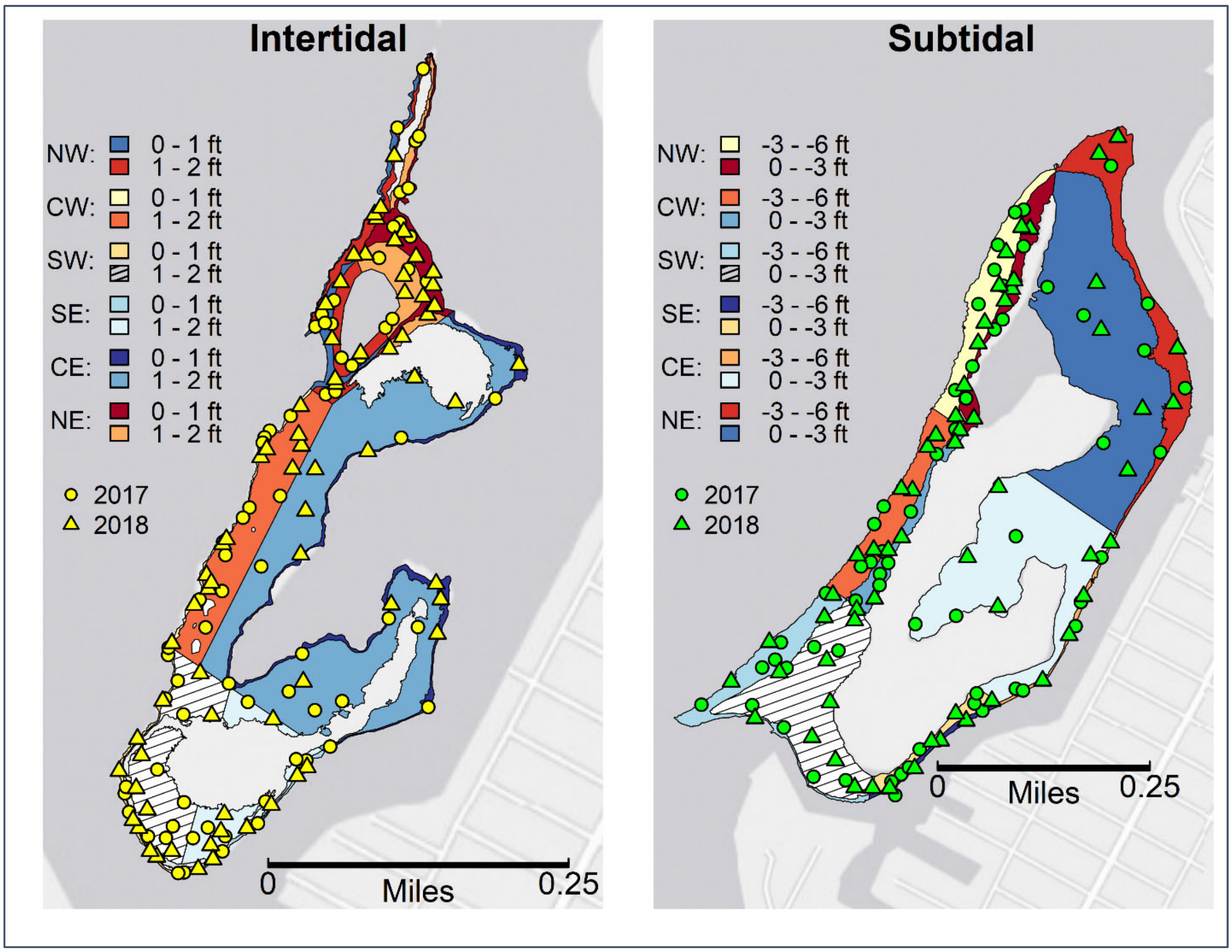

Figure 2. Maps showing intertidal (left) and subtidal (right) elevation or depth strata and point locations sampled in 2017 and 2018. (NW=northwest, CW=central-west, SW=southwest, SE=southeast, CE=central-east, NE=northeast).

2.1.1. Intertidal

At each of the intertidal sampling points vegetative parameters (species composition, percent cover of each species, and total standing biomass of *Spartina alterniflora*) were evaluated within a 1 m^2 quadrat; a single soil core (3 cm diameter by 10 cm deep) was collected for analysis of bulk density and soil total organic carbon (TOC) content; and a porewater sample (10 cm depth) was collected for analysis of hydrogen sulfide concentration. Vegetative percent cover was categorized according to the Carolina Vegetation Survey (CVS) method (Peet et al. 1998) which involves classifying the percent cover of each species in a one m^2 area according to 10 pre-defined categories (Table 1). At each sampling point where *S. alterniflora* was present the maximum green leaf height was documented for 10 *S. alterniflora* stems randomly chosen as the first 10 stems that intersected a string running across the centerline of the quadrat. In addition to measuring heights, we counted the total number of *S. alterniflora* stems in one quarter of the quadrat (0.25 m^2). Average plant height (average of the 10 measured stems) and total stem count were used to estimate total standing *S. alterniflora* biomass based on previously identified allometric relationships (Davis et al. 2017).

Table 1. Percent cover categories used in the Carolina Vegetation Survey (Peet et al. 1998).

CVS Category	Vegetative Cover
1	< 3 stems
2	0.1 - 1%
3	1 - 2%
4	2 - 5%
5	5 - 10%
6	10 - 25%
7	25 - 50%
8	50 - 75%
9	75 - 95%
10	95 - 100%

Sediment cores were dried at 60 ^oC for 48 hrs and then weighed for calculation of bulk density. Once dry, the entire core was homogenized and subsampled in duplicate for analysis of TOC using a Costech ECS 4010 elemental analyzer. Porewater samples were collected with a peristaltic pump and push-point sampling device, preserved in zinc acetate immediately upon collection, and analyzed within one month of collection using Cline's method (Cline 1968). The elevation of each sample point was determined with Real Time Kinematic-Global Positioning System (RTK-GPS; Trimble® 5800 and R6-3 receivers) and referenced to the National Geodetic Survey mark 15 W 3 (NGS PID DM5212) located at the west end of Berkley Ave, Beach Haven, NJ. The mark is a disc set in concrete. All fixed vegetation plot locations were collected as 5-second topographic points.

The plantings exhibited varied success in the early years. By 2018 the intertidal region of the placement area was characterized by bare sediment dotted with sparse clumps of *S. alterniflora* (Figure 3). As a result, there was no vegetation present at any of the randomly selected sample points within the placement area in 2017 or 2018. To better characterize the vegetation present in the placement area, we modified our sampling approach in 2019 to target the zone where *S. alterniflora* was beginning to thrive. We established five transects that radiated outward from the central high-elevation mound, beginning at the upper edge of *S. alterniflora* growth and extending to the lower edge of vegetation or the shoreline, whichever came first (

Figure 4). Sample plots were located at equally spaced intervals across the extent of each transect (five plots per transect); total transect length ranged between 20 and 70 m. Sampling at each plot involved the same vegetative and soil parameters described above for point-based sampling in 2017 and 2018.

Figure 3. The placement area in 2018 was characterized by patches of vegetation scattered across a mostly unvegetated landscape.

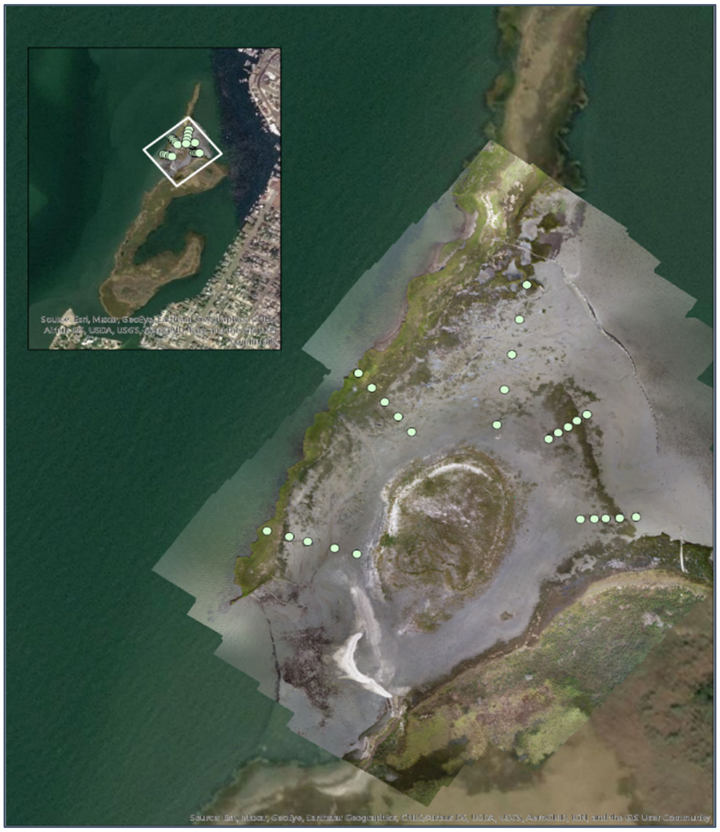

Figure 4. Aerial imagery of the placement region with 2019 vegetation sample points. Five new transects were established to characterize *S. alterniflora* within the created marsh. Green dots indicate sampling points along each transect.

2.1.2. Subtidal

At each sampling point, water quality parameters (salinity, temperature, dissolved oxygen content, and pH) were measured with a hand-held YSI 556 multiprobe instrument. Sediment samples for analysis of sediment grain size, TOC, and macrobenthic infauna community composition were collected using a Young-modified Van Veen grab sampler (0.04 m^2). Due to time constraints and the large number of sampling sites (60), we obtained only one replicate for infaunal analysis. Sediment grain size and TOC samples were placed in jars on site and kept cold until return to the lab for further processing. Infauna samples were sieved on a 0.5 mm mesh screen, preserved in 10 % buffered formalin in the field, and later transferred to 70 % ethanol in the lab. Analyses of infaunal communities were carried out in R (R Core Team 2020) using the *stats*, *emmeans*, and *vegan* packages. Sediment grain size samples were analyzed for moisture content and percent silt+clay following the rough analysis in Plumb (1981). TOC was measured using the same method as the intertidal samples, described above. Benthic infaunal samples were sorted into major taxonomic categories, further identified to the lowest possible taxonomic level, and enumerated using a dissecting microscope. In 2018 we added an analysis of benthic vegetative cover to our subtidal point-based sampling regime. Methods for benthic vegetation surveys are described in detail in section 2.2: Submerged Aquatic Vegetation.

2.2. Submerged Aquatic Vegetation and Benthic Macroalgae

In response to the stated interest of partner agencies (U.S. Fish and Wildlife Service and USACE Philadelphia District) in documenting the occurrence of Submerged Aquatic Vegetation (SAV) around Mordecai Island, and particularly along the western shoreline north of the placement area, we modified our sampling approaches in 2018 and 2019 to capture data on SAV distribution. In 2018, we used a drop-camera approach to document benthic cover at all of the subtidal sampling stations. Underwater photographic images were captured with a GoPro Hero 4® camera mounted to a PVC sampling frame that consisted of a vertical pole attached to a square quadrat (0.5 m on a side; Figure 5a). At each station, the frame was lowered to the bottom from the same location on the boat until the quadrat sat firmly on the sediment surface. The GoPro collected images at 5-second intervals while the frame was lowered into the water and retrieved. Because there were multiple images collected at each station, only the highest resolution image was selected for analysis. Analysis involved estimation of the percent cover of SAV and/or benthic macroalgae within the area of the quadrat at each station. Image quality was impacted by a number of factors including water depth and clarity, intensity of sunlight, and steadiness of the camera (waves and currents made this a challenge). Adobe Lightroom® was used to enhance image quality by adjusting brightness and contrast. In some photos, even after enhancement, the type of vegetation present could not be determined conclusively. In these cases the cover type was recorded as "undetermined".

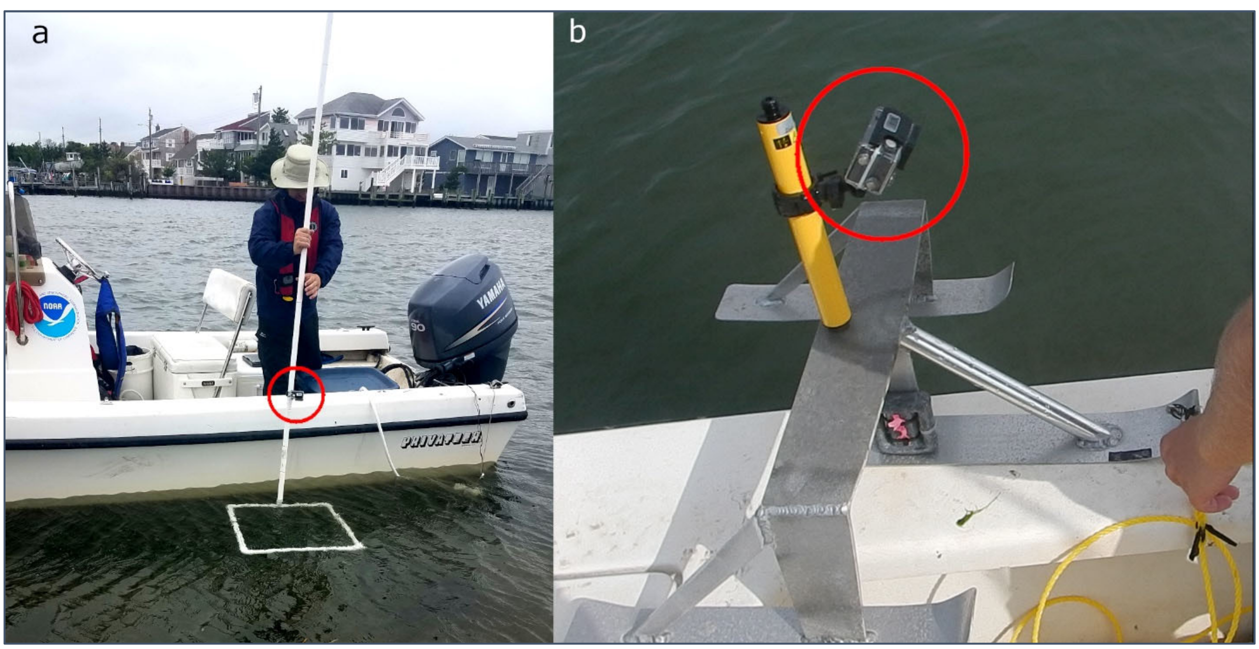

Figure 5. Submerged aquatic vegetation (SAV) data collection: a) SAV sampling frame with attached GoPro (in red circle), used for point-based sampling in 2018; b) SAV sled with attached GoPro, used for transect sampling in 2019.

In 2019, SAV quantification efforts were concentrated in the nearshore waters to the west of the island. At randomly selected locations, an aluminum sled with attached GoPro camera (Hero4; Figure 5b) was deployed and the towing line was payed out while the boat was moved approximately 30 m to the opposite end of the transect. The locations of the beginning and end of each transect were recorded with RTK-GPS. The line was retrieved by hand while pulling the sled across the substrate at a constant speed. During the tow, the camera collected time-stamped imagery at 5-second intervals. The time stamps on the initial and final photos were used to determine the total tow time for each transect. The imagery from each transect was subsampled at constant intervals to collect 6 distinct frames (no overlap in coverage between frames) for analysis of vegetative cover. Using total tow time and order of image collection, it was possible to estimate the distance along the transect at which each image was collected. Percent cover of SAV and macroalgae was estimated by visual analysis of photos. The entire frame of each photo was included in each analysis. During some of the deployments the camera angle was inadvertently altered resulting in a slightly different camera orientation and therefore field of view. The total benthic surface area captured by images ranged from 0.5 to 1.0 m^2.

2.3. Real Time Kinematic Surveys

In all three sample years (2017, 2018, and 2019), RTK-GPS surveys (NOAA 2013) were conducted along the length of the western shoreline to document short-term changes in shoreline position. Additional surveys were conducted in three shoreline-proximal areas of interest (AOIs): the sediment placement area, the unmodified central expanse of the western shoreline (referred to here as “natural”), and the southern region that is stabilized with geotubes (Figure 6a) for the purpose of comparing changes in modified (geotube and placement) and unmodified (natural) areas. Surveys involved mapping shoreline position (defined as the maximum western extent of vegetation) and creating digital elevation models of the marsh platform adjacent to the shoreline in the three AOIs. In 2018 and 2019, additional surveys were conducted to establish the elevation profile of the full extent of the placement area. These surveys involved making multiple passes across the central mound in a crisscross pattern. The RTK-GPS base station was located on bench mark 15 W 3 in Beach Haven during all surveys. All AOI and shoreline surveys were conducted with a GPS rover attached to a modified bicycle which was used to move over the surface at a steady walking rate (Figure 6b). The rover unit collected position data at 3-second intervals in continuous topo mode. Points were collected cumulatively with an average spacing of 1 m. Before and after RTK-GPS surveys, the rover was positioned on a Class B mark (steel rod driven to the point of refusal that was installed on the island by NCCOS researchers in 2017) which served as a fixed reference point for determination of the horizontal and vertical accuracy of measurements. During all occupations on the Class B mark there was < 0.01 m orthometric difference and no vertical adjustments were made. The uncertainty (horizontal) was 0.1 m based on 3x the Horizontal Dilution of Precision (HDoP) during the RTK-GPS surveys. All field-collected GPS data were imported to Trimble Business Center for post-processing then exported to ArcGIS for visualization and analysis.

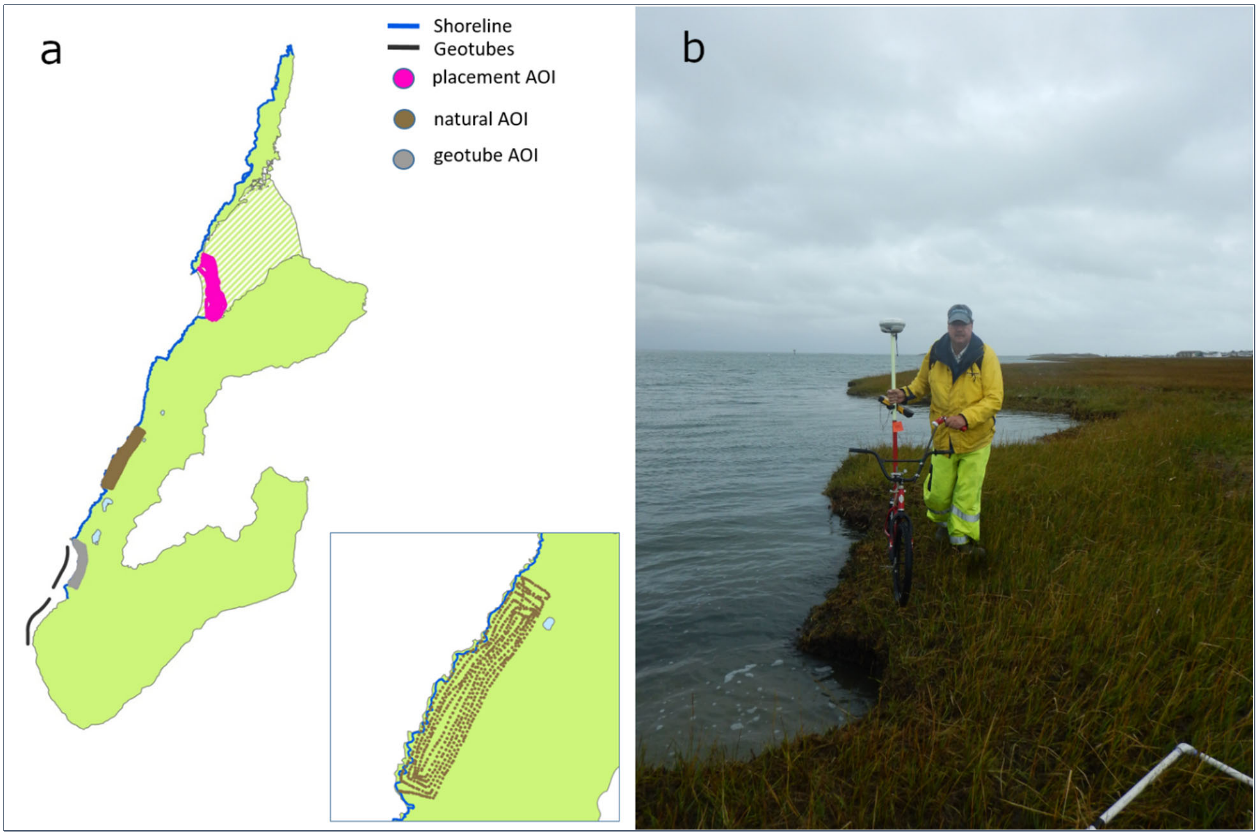

Figure 6. RTK shoreline surveys: a) three shoreline areas of interest – placement, natural, and geotube (inset = close-up of points in natural AOI to demonstrate spacing); b) bike-mounted rover setup.

Changes in shoreline position over time were calculated with the Digital Shoreline Analysis System (DSAS) v 5.0 add-in for ArcGIS 10.7 (Himmelstoss et al. 2018). DSAS uses a series of shoreline shapefiles with known collection dates to calculate the average rate of change in shoreline position over time. This is accomplished by generating multiple transects perpendicular to the shoreline that extend from a user-defined baseline through each successive shoreline. The distance between the baseline and successive shorelines is measured along each transect and then averaged among all transects to estimate an overall rate of shoreline change. DSAS incorporates a user-defined positional uncertainty for each shoreline. Change in shoreline position since 1970 was analyzed for the entire length of the western shoreline with 10 m spacing between transects (Figure 7). In addition to the field-collected data, we used historical shoreline position to investigate long-term rates of change. Historical shoreline positions (1970, 1977, and 2007) were generated by hand-digitizing the shoreline boundary from aerial imagery available from the New Jersey Department of Environmental Protection Geographic Information System. All three image sets are available as natural color ortho-photographs produced at a scale of 1:2400 with an average horizontal accuracy of ±1.2 m at a 95% confidence level.

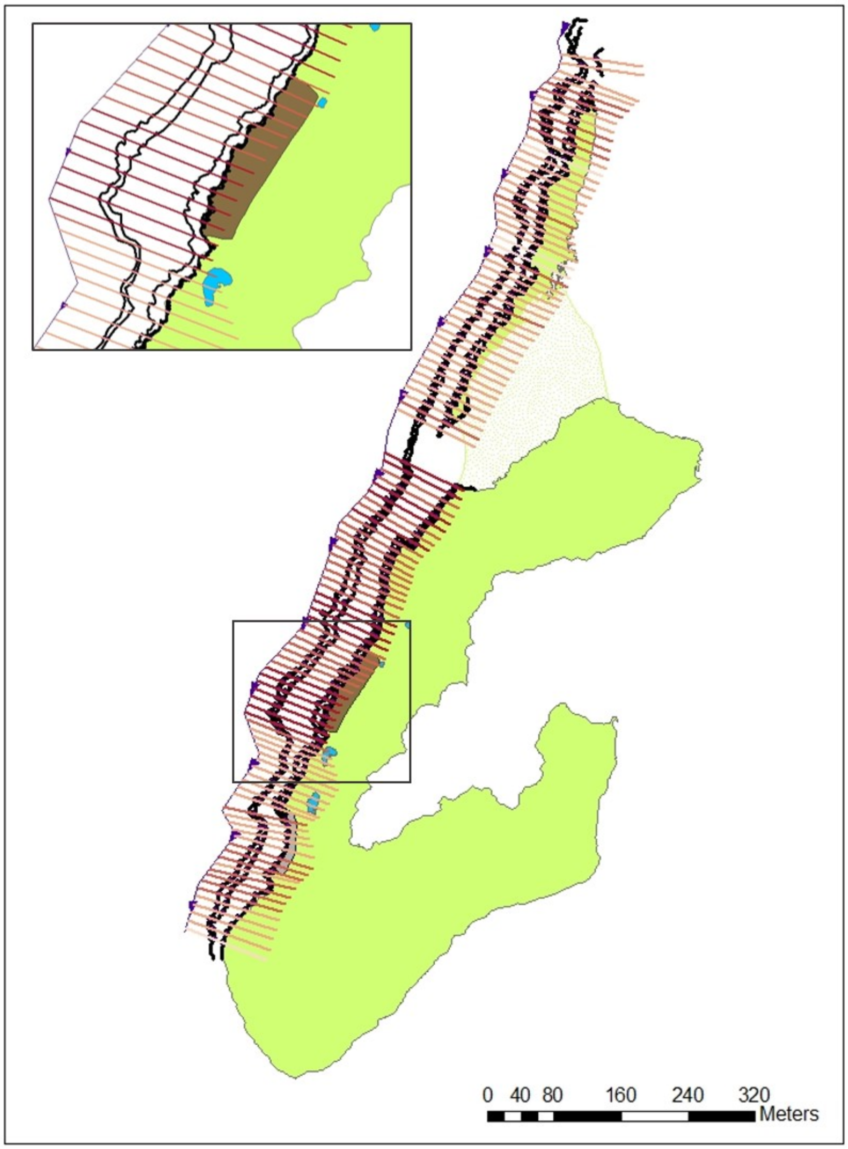

Figure 7. DSAS analysis of change in shoreline position since 1970 along the western shoreline of Mordecai Island. Shore-parallel lines represent historical shoreline positions (1970, 1977, 2017, and 2019). Shore perpendicular lines represent transects along which change in shoreline position was measured.

Digital Elevation Models (DEMs) were constructed for each of the shoreline-adjacent AOIs using the Topo-to-Raster tool in ArcMap to interpolate elevation data collected during the RTK-GPS surveys. Additional DEMs were created for the entire placement area in 2018 and 2019 by merging all RTK-GPS data collected within the placement region for that year (shoreline surveys, mound surveys, and vegetation sampling points) and interpolating elevation across all points in the merged data set. In 2017, RTK-GPS surveys were not conducted across the central mound region of the placement area due to the presence of an extensive network of goose exclusion fencing. The 2017 DEM was created from bare earth Lidar collected by the USACE National Coastal Mapping Program between August 15 and September 9, 2017. The spatial resolution of the Lidar data was 1 m.

2.4. Unmanned Aerial Systems (UAS) Surveys

In 2019 we conducted Unmanned Aerial Systems (UAS) surveys of the island to supplement ground-based efforts to document elevations and vegetative distributions within the placement area and nearshore subtidal vegetation. All UAS surveys were conducted with a DJI Phantom 4 Pro® equipped with a 20 MP CMOS sensor. Surveys involved pre-planned transects including one survey designed to capture the entire island and a second, higher-resolution survey of the placement area. The whole-island survey was flown at an altitude of 300 ft with 75 % front overlap and 65 % side overlap resulting in 380 images. The placement area transect was flown at 100 ft with 75 % front and 70 % side overlap resulting in 231 images. A neutral density polarizing filter (ND8-PL) was used for all flights to minimize glare. All images were processed in Agisoft Metashape (v 1.6). Orthophotomosaics of the placement area had a ground resolution of 8 mm/pixel, while those of the whole island survey had a ground resolution of 2.4 cm/pixel. To optimize image alignment of the placement region data set, 19 ground control points surveyed with RTK-GPS were used to establish their precise locations. The resulting orthophotomosaics provide a visual record of the extent of nearshore subtidal vegetation and colonization of the placement area as of 2019.

3. Results and Discussion

3.1. Intertidal Vegetation and Sediments

The lower intertidal vegetative community of Mordecai Island was dominated by *S. alterniflora*, which was the most abundant form of vegetation present at elevations between -0.03 and 0.3 m relative to the North American Vertical Datum of 1988 (NAVD88). In some cases, particularly where mussels were abundant in the marsh substrate, bladderwrack (*Fucus vesiculosis*) was present within the *S. alterniflora* stands. A mixed community of *S. alterniflora*, *Spartina patens, Distichlis spicata, Salicornia sp.*, and *Limonium sp*. dominated at higher intertidal elevations (Table 2; Figure 8). Our sampling strategy was intentionally focused on intertidal elevations and therefore did not allow for a comprehensive list of all vegetative species present on Mordecai Island.

Table 2. Elevation range over which each species was observed in randomly selected sample points in 2017 and 2018.

Species	Elevation range of occurrence (m NAVD88)	Occurrence (out of 131 total plots sampled)
Spartina. alterniflora	-0.03 – 0.63	101
Distichlis spicata	0.35 – 0.67	17
Salicornia sp.	0.25 – 0.67	34
Spartina. patens	0.4 – 0.67	16
Limonium sp.	0.35 – 0.58	17
Fucus vesiculosus	-0.03 – 0.28	8
Phragmites australis.	0.56 – 0.67	2
Unvegetated	0.06 – 0.69	19

At elevations greater than 0.7 m NAVD88, *Phragmites australis* was abundant and woody shrubs dominated above the *P. australis* zone. The occurrence and abundance of vegetative species were consistent between 2017 and 2018 with the exception of there being a greater number of plots with sparse coverage of *Limonium* and *Salicornia* in 2018 (Figure 8; CVS values of 1 or 2). During our 2018 sampling effort, *Salicornia* appeared to be in peak senescence based on the widespread occurrence of stems that were bright yellow and red (Figure 9). We hypothesize that the bright coloration made *Salicornia* more visible in 2018 and thus increased the likelihood of detection in plots where it was rare. Mean stem heights, stem counts, and total standing biomass of *S. alterniflora* were similar among the two years (Table 3). These variables did not differ significantly by region (North, Central, South) or by shoreline face (East, West).

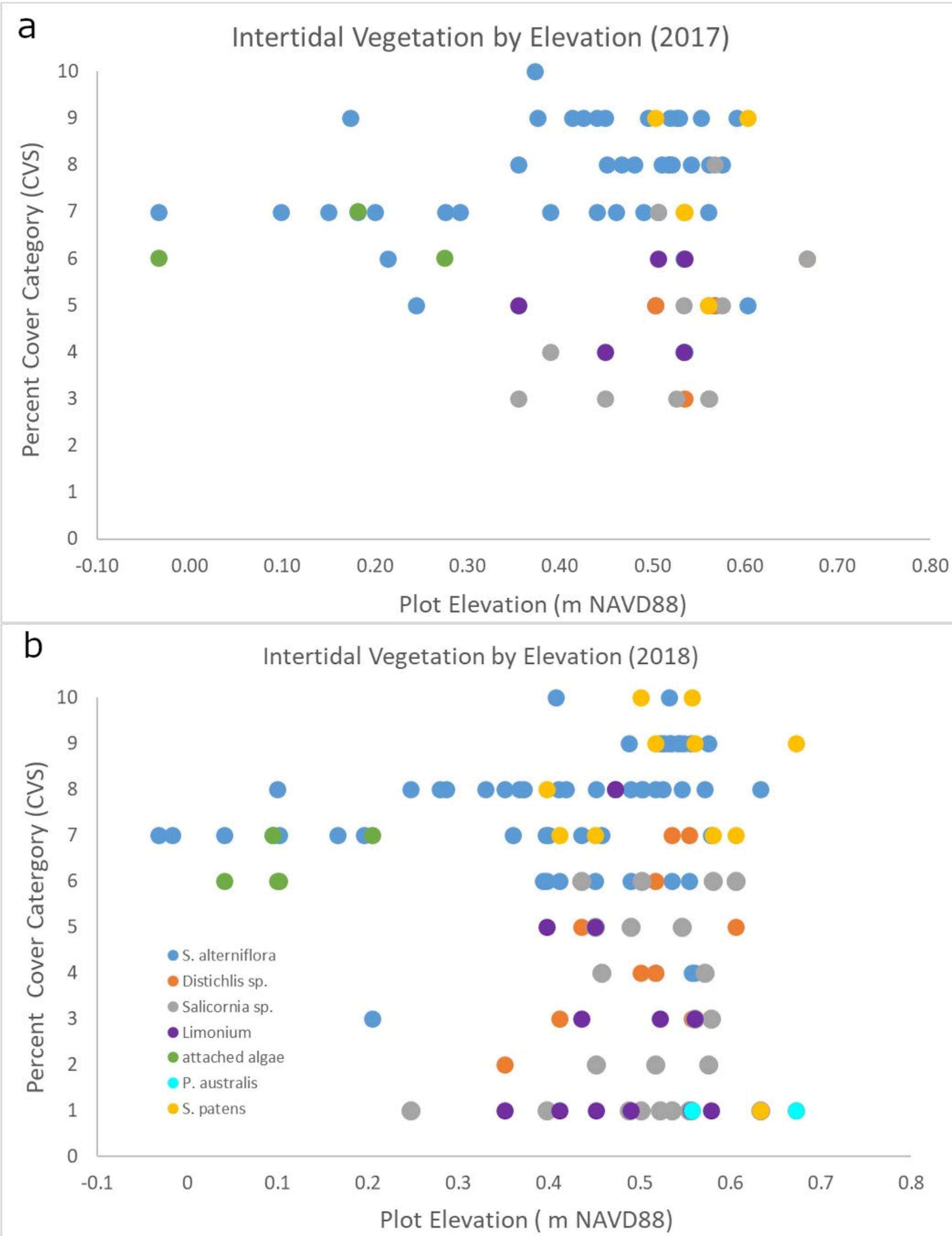

Figure 8. Percent cover of intertidal vegetation by species across all intertidal sampling points in a) 2017, and b) 2018. Percent cover scores represent categories defined by the Carolina Vegetation Survey (Peet et al. 1998).

Figure 9. Senescent *Salicornia* sp. are easily distinguished from the surrounding vegetation based on their bright coloration.

Table 3. Occurrence, height, stem count, and standing live biomass of *S. alterniflora* by year. Data from 2017 and 2018 represent all vegetation sample points where *S. alterniflora* was present. None of the sample points within the placement region were vegetated in either year. Data from 2019 represent transect based sample points collected only in *S. alterniflora* vegetated regions of the placement area. N = total number of plots in which *S. alterniflora* was present.

		Stem Height (cm)			Stem Density (#/m^2)			Biomass (g/m^2)		
Year	N	Min	Max	Mean (SE)	Min	Max	Mean (SE)	Min	Max	Mean (SE)
2017	45	10	99	29 (2.6)	47	4704	723 (120)	21	3006	516 (93)
2018	56	9	75	29 (1.9)	12	3088	768 (99)	11	1495	455 (49)
2019	24	15	51	26 (2.0)	6	1080	245 (56)	2	410	157 (32)

There was no vegetation present at any of the randomly selected sample points that fell within the sediment placement area in 2017 or 2018. The original planting scheme involved planting individual plugs in a grid with 1-foot spacing between planting units. Visual observation of the area indicated that some of the planted *S. alterniflora* in the lower elevation regions surrounding the mound (< 0.4 m NAVD88) had become established and was beginning to spread. However, as of September 2019, a large swath surrounding the central mound at elevations of 0.4 to 0.85 m NAVD88 remained unvegetated. In order to characterize the intertidal vegetation present in the placement region and to provide a reference for future evaluations of planting success at this site, we modified our approach in 2019 by intentionally placing transects in regions where *S. alterniflora* was beginning to become established (Figure 4). Comparison of the vegetative community along these transects with that of the natural regions of the island (Figure 10 and Figure 8, respectively; Table 3) shows that while average stem heights of *S. alterniflora* in the placement region are similar to those of natural regions, the placement region lags behind the natural *S. alterniflora* communities in stem density, biomass, and species diversity.

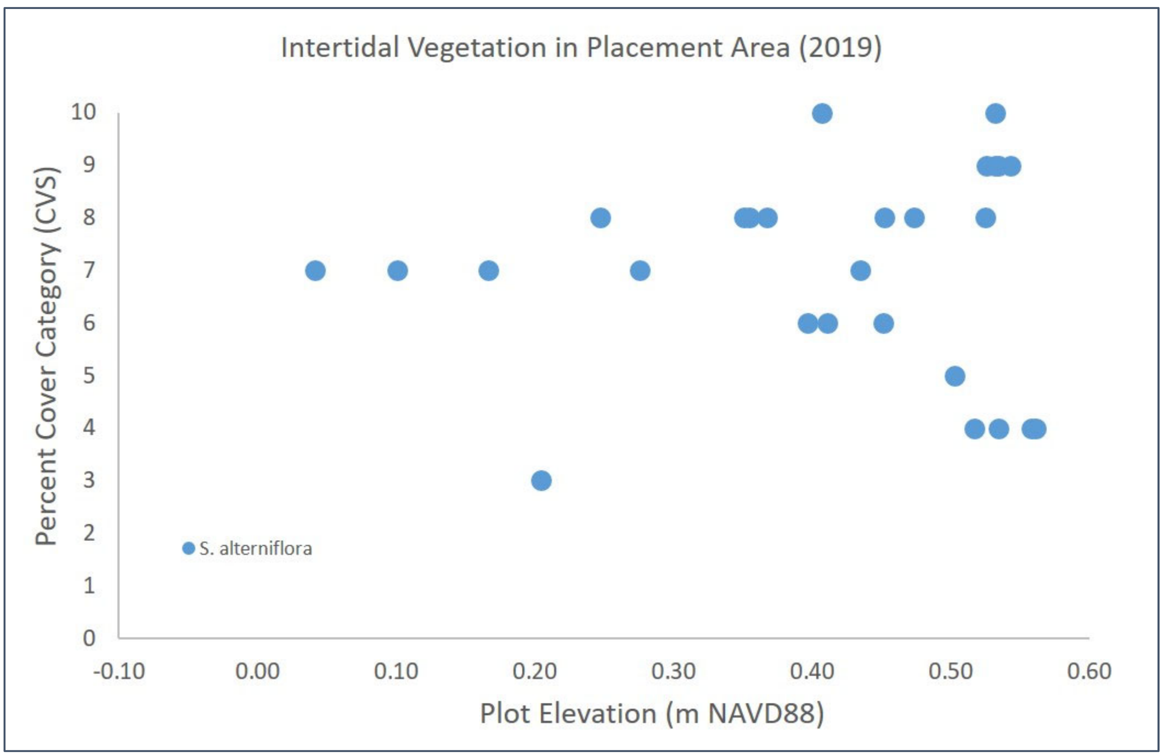

Figure 10. Percent cover of intertidal vegetation in transect-based sample plots in 2019 (*Spartina alterniflora* was the only species present).

Organic carbon tends to increase overtime in wetland soils due to the accumulation of dead and decaying plant root tissue which breaks down slowly in saturated sediments. Wetland soils with higher organic content tend to hold more water than sandy soils, and thus experience less drying between tidal cycles. The slow decay of organic material releases nutrients that are crucial for fueling plant growth. The TOC content of soils is thus an indicator of soil quality, and previous research has demonstrated that low organic matter content can limit plant growth in restored wetlands (Sutton-Grier et al. 2009). Sediment TOC content in all vegetation plot samples in 2017 and 2018 ranged from below detection to a maximum of 24 % TOC by weight with an overall average of 5.3 % and did not vary significantly by year. TOC content was significantly greater in the 1 to 2 ft sampling stratum than in the lower stratum in the Central and South regions only ($F_{5,134}$ = 11.1, p = <0.0001; Table 4). The entire sediment placement area fell within the North region and many samples from within the placement area did not contain detectable amounts of TOC. As a result, the average % TOC value for the North region was significantly lower than samples from similar elevations in the Central and South. In 2019, when samples were only collected within the placement region, sediment TOC content averaged 0.45 % and ranged from 0.1 to 1.5 %. Sediment bulk density was inversely correlated with TOC content (Figure 11).

Table 4. Sediment total organic carbon content (2017 and 2018) from vegetation sampling points averaged by elevation stratum and sample region.

	Sediment % Total Organic Carbon by Weight: Mean (SE)		
Elevation Stratum	North	Central	South
0 – 1 ft.	1.7 (0.6)	5 (1.3)	3.3 (1.0)
1 – 2 ft.	2.0 (0.8)	8.9 (1.7)	11.5 (1.5)

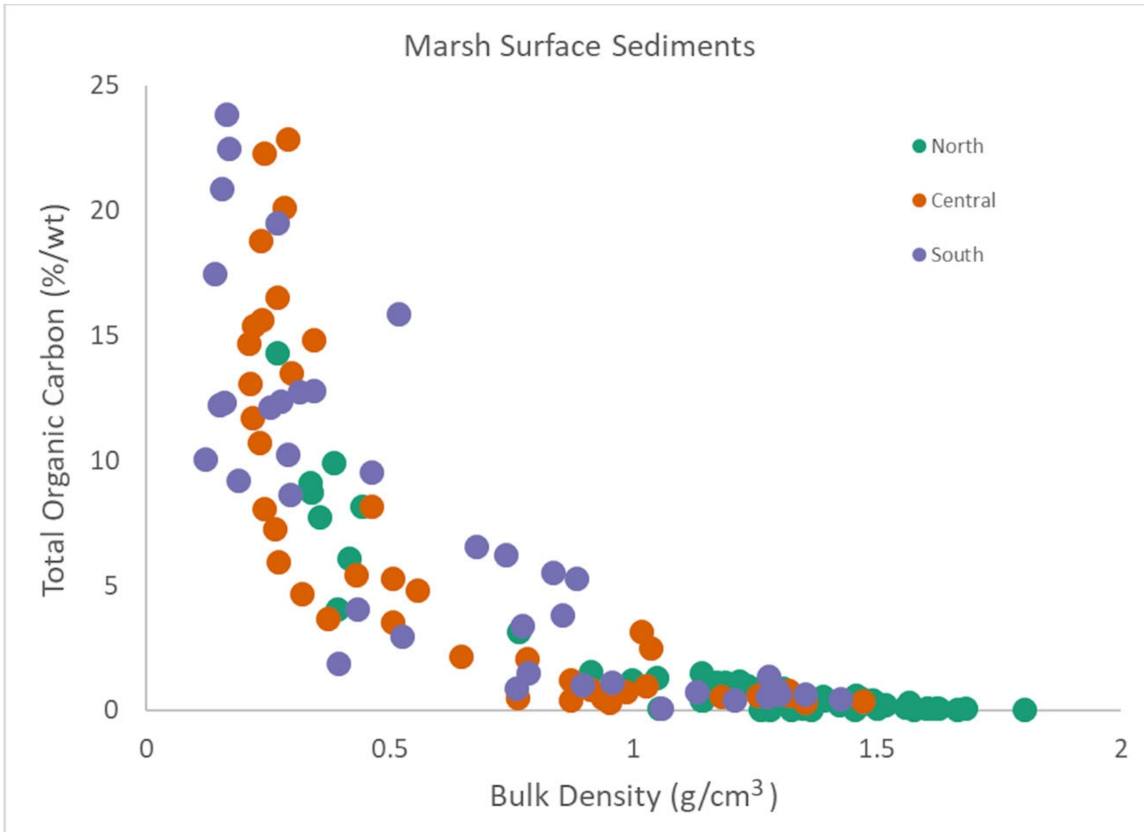

Figure 11. Sediment total organic carbon (TOC) content as a function of bulk density based on all samples in all years. The low TOC, high bulk density samples in the North sampling region represent conditions within the placement region.

Sediments within the placement region are, on average, denser and lower in TOC than those of the surrounding marsh. Previous efforts to determine the rate of TOC accumulation in newly created marshes suggest that it will take decades before sediment TOC content in the placement region reaches equivalence with native marsh sediments (Davis et al. 2015). It is worth noting that high bulk density (> 1 g/cm^3), low TOC content (< 2 % TOC by weight) sediments were detected at multiple sample points within the Central and Southern regions, thus conditions within the placement area are not outside of those that occur naturally on Mordecai (Figure 11).

Hydrogen sulfide (H_2S) is a by-product of sulfate reduction, the microbial oxidation of organic compounds using sulfate as a terminal electron acceptor. Sulfate reduction is often the dominant respiratory pathway in marine and estuarine wetland sediments because oxygen is quickly depleted in inundated sediments, and sulfate is abundant in seawater. Hydrogen sulfide has phytotoxic effects on plants and at high concentrations (> 3 mM) has been shown to interfere with the ability of wetland plants to take up inorganic nitrogen (Lamers et al. 2013, Bradley and Morris, 1990). Thus, elevated H_2S concentrations can indicate sub-optimal conditions for vegetative growth. Porewater sulfide concentrations were greater in samples from the lower elevation stratum, likely as a result of longer inundation times, but did not differ significantly by region (Table 5), suggesting that sediment geochemistry within the root zone in the placed sediments was similar to that of the native marsh. While average concentrations in lower elevation samples were near the 3 mM range, these values are within the range of H_2S concentrations previously reported in other salt marsh porewaters (Koretsky et al. 2005, Koretsky and Miller 2008) and did not appear to be negatively impacting vegetative growth in the natural regions of Mordecai.

Table 5. Concentrations of hydrogen sulfide in porewater at vegetation sample points in 2017 and 2018 averaged by elevation and sample region.

	Porewater H_2S Concentration (µM): Mean (SE)		
Elevation Stratum	North	Central	South
0 – 1 ft.	2,086 (582)	3,009 (582)	2,276 (545)
1 – 2 ft.	1,207 (582)	1,632 (528)	1,393 (562)

Previous efforts to characterize the pace of vegetative colonization in planted, created *S. alterniflora* marshes have shown that it can take 3 to 10 years before they reach equivalence with their natural counterparts in terms of vegetative cover (Craft et al. 2003, Currin et al. 2008). The pace at which recolonization occurs is likely impacted by environmental characteristics like extreme weather events, nutrient availability and herbivory pressure in addition to site physical characteristics including position within the tidal frame, and planting density among other factors. In 2017, large swaths of the planted area on Mordecai were devoid of vegetation, indicating that many of the original plantings did not survive. By 2019, *S. alterniflora* was beginning to spread laterally into the placement region from the adjacent marsh but there was still a large swath of unvegetated sediment, spanning 0.4 to 0.6 m NAVD88 in elevation, surrounding the central mound. We anticipate that with favorable environmental conditions, *S. alterniflora* will continue to spread laterally within the placement region and colonize the bare areas that are within its elevation growth ranges. As of 2019, the highest regions of the placement area (elevations > 0.8 m NAVD88), which were intended to remain unvegetated to provide nesting habitat, had become heavily colonized with *P. australis*, an invasive plant that thrives in well-drained soils (Bart and Hartman 2000).

3.2. Subtidal Sediments and Benthic Infauna

Sediment percent silt+clay and TOC content showed clear differences between the western and eastern portions of the subtidal sampling area in 2017 and 2018, with sites along the eastern, more sheltered part of the island typically having higher silt+clay and TOC content than those adjacent to the more exposed western shoreline (Figure 12). In both years, percent silt+clay and TOC were highly correlated ($p < 0.001$, $R^2 \geq 0.94$), as illustrated in Figure 13.

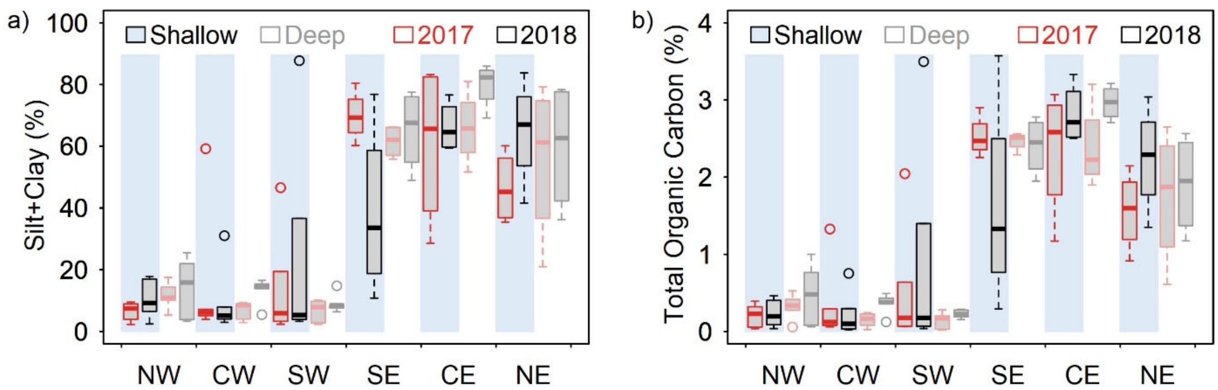

Figure 12. Ranges in percent silt+clay and total organic carbon (TOC) content measured in sediments sampled from subtidal sites around Mordecai Island in 2017 and 2018. Abbreviations along the x-axis refer to regions of the island (NW=northwest, CW=central west, SW=southwest, SE=southeast, CE=central east, NE=northeast).

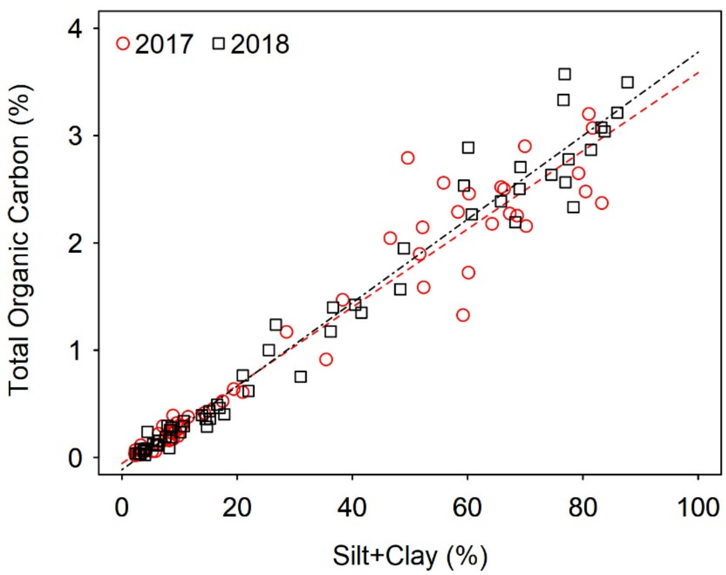

Figure 13. Relationship between percent silt+clay and total organic carbon (TOC) in subtidal sediment samples from Mordecai Island in 2017 and 2018.

Overall measures of infaunal species richness (# of taxa), diversity (Shannon H'), and density (# of individuals/m^2) were similar between stations sampled in 2017 and 2018 (Table 6). Factorial analysis of variance (ANOVA) did not find any significant difference in infaunal richness between years. The analysis indicated a significant effect of side (west vs. east), but this effect varied by depth zone (Figure 14; side, $F_{1,96} = 8.34$, $p = 0.005$; side:depth interaction, $F_{1,96} = 8.48$, $p = 0.004$). Due to the significant interaction of side and depth, separate ANOVAs by depth zone were conducted. Among shallow sites, species richness was significantly higher in the western part of Mordecai Island than the east ($F_{1,48} = 16.638$, $p < 0.001$). No significant difference was observed for deep sites.

Table 6. Distributional properties (mean, standard error (SE), range) of benthic metrics calculated for infaunal samples collected from 120 stations in 2017 and 2018.

	2017 (n=60)		2018 (n=60)	
Benthic Metric	Mean (SE)	Min – Max	Mean (SE)	Min – Max
Richness (# of taxa)	23 (0.8)	11 – 40	24 (0.8)	11 – 39
Diversity (H')	2.8 (0.1)	1.2 – 4.3	3.1 (0.1)	1.3 – 4.3
Density (#/m^2)	10,636 (1,123.7)	1,025 – 34,200	8,102 (666.3)	2,200 – 26,100

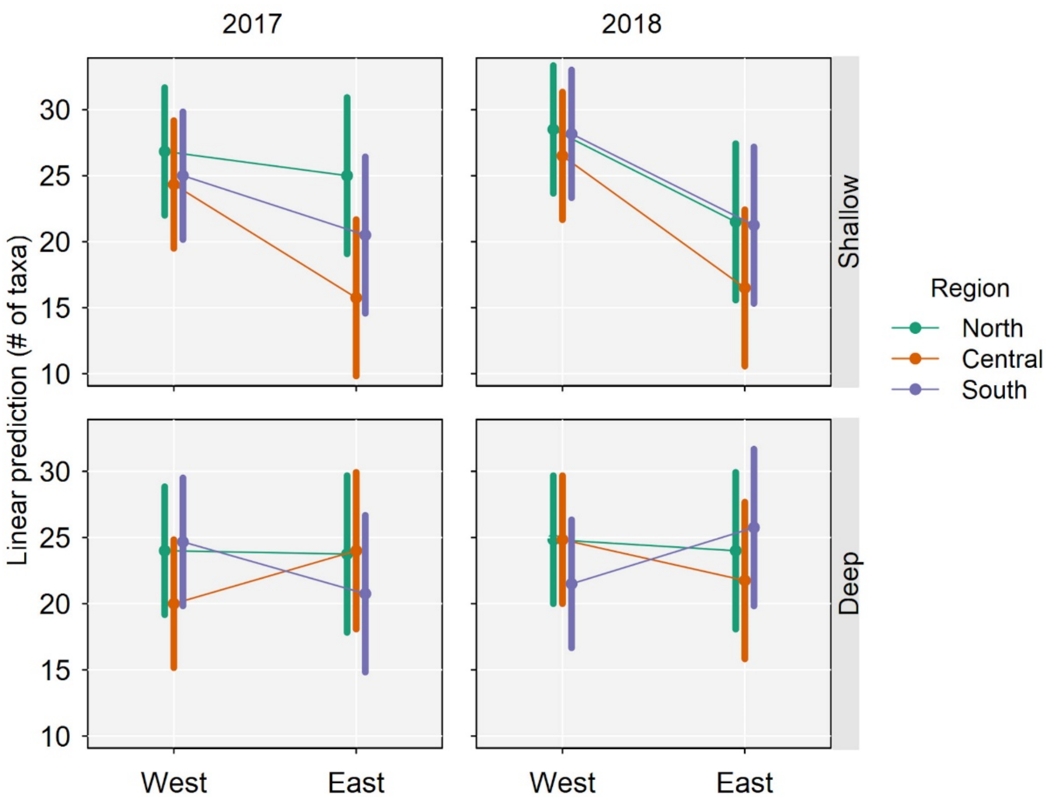

Figure 14. Estimated marginal means (and 95% confidence intervals) obtained from a linear model relating species richness (# of taxa) to year, region, side, and depth zone.

With respect to infaunal diversity (Shannon H’), there were significant pair-wise or higher-order interactions; hence, separate ANOVAs by depth zone and year were used. For shallow sites in both years (2017 and 2018; Figure 15, diversity was higher at stations on the western side of Mordecai Island (2017, side, $F_{1,24}$ = 13.81, p = 0.001; 2018, side, $F_{1,24}$ = 37.22, p < 0.001); in 2017, diversity was also highest in the northern region irrespective of side (region, $F_{2,24}$ = 5.61, p = 0.010). For deep sites, diversity also tended to be higher at western sites, but this varied by region (greater difference in central region in 2018) and significance (difference between west and east not significant in 2017, p = 0.057).

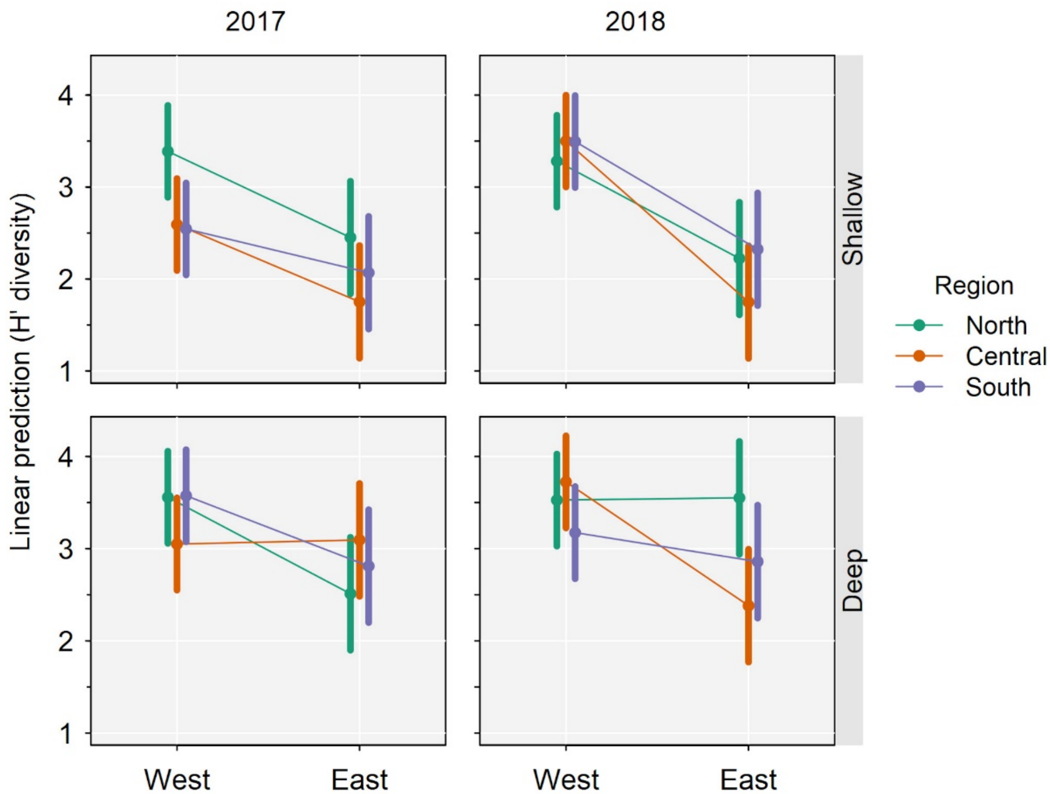

Figure 15. Estimated marginal means (and 95% confidence intervals) obtained from a linear model relating species diversity (Shannon H’) to year, region, side, and depth.

For infaunal density (#/m^2), there was a highly significant effect of side and depth zone (Figure 16), but due to the presence of significant interactions with year and region, ANOVAs were conducted separately by year and depth zone. Whereas infaunal richness and diversity were higher for sites on the western side of Mordecai Island, the reverse was true of density. At shallow sites in both years, density was higher at eastern sites (2017, side, $F_{1,24}$ = 13.54, p = 0.001; 2018, side $F_{1,24}$ = 5.76, p = 0.024), though in 2017 the difference was greatest in the northern and central regions. Also at shallow sites, densities were higher overall (irrespective of side) in the southern region in 2017 (region, $F_{2,24}$ = 4.90, p = 0.016). At deeper sites in 2017, densities were higher at eastern stations, mainly in the northern region (side, $F_{1,24}$ = 7.08, p = 0.014; region:side interaction, $F_{2,24}$ = 3.26, p = 0.056).

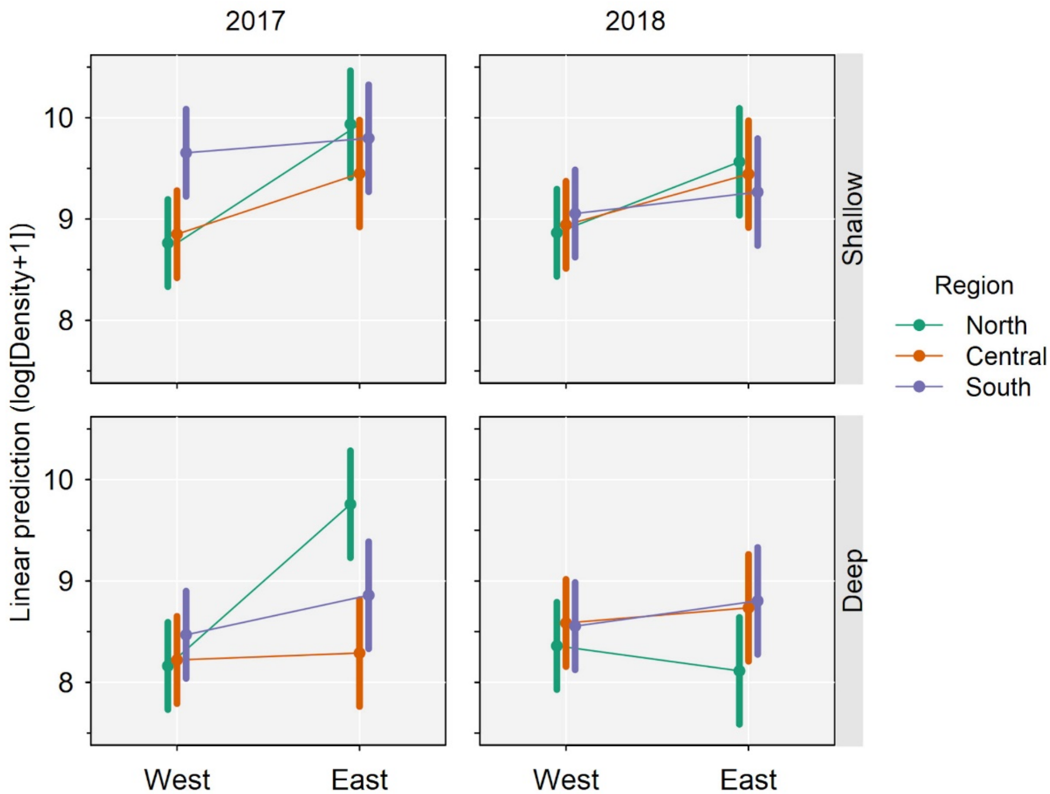

Figure 16. Estimated marginal means (and 95% confidence intervals) obtained from a linear model relating species density (log-transformed # of individuals per m^2) to year, region, side, and depth zone.

Results of hierarchical cluster analysis (using Bray-Curtis dissimilarities, unweighted pair group method with arithmetic mean) and non-metric multidimensional scaling (NMDS) of square-root transformed species abundance revealed patterns in the distribution of infaunal assemblages that were in agreement with analysis of benthic metrics (species richness, diversity, and density). In 2018, stations separated mainly on the basis of percent silt+clay and TOC. The larger site groups in Figure 17 correspond primarily to sites having low vs. high percent silt+clay and TOC in the western vs. eastern part of the island, respectively. These patterns are further illustrated by the results of NMDS (Figure 18), with percent silt+clay and TOC driving the separation of sites along the first ordination axis.

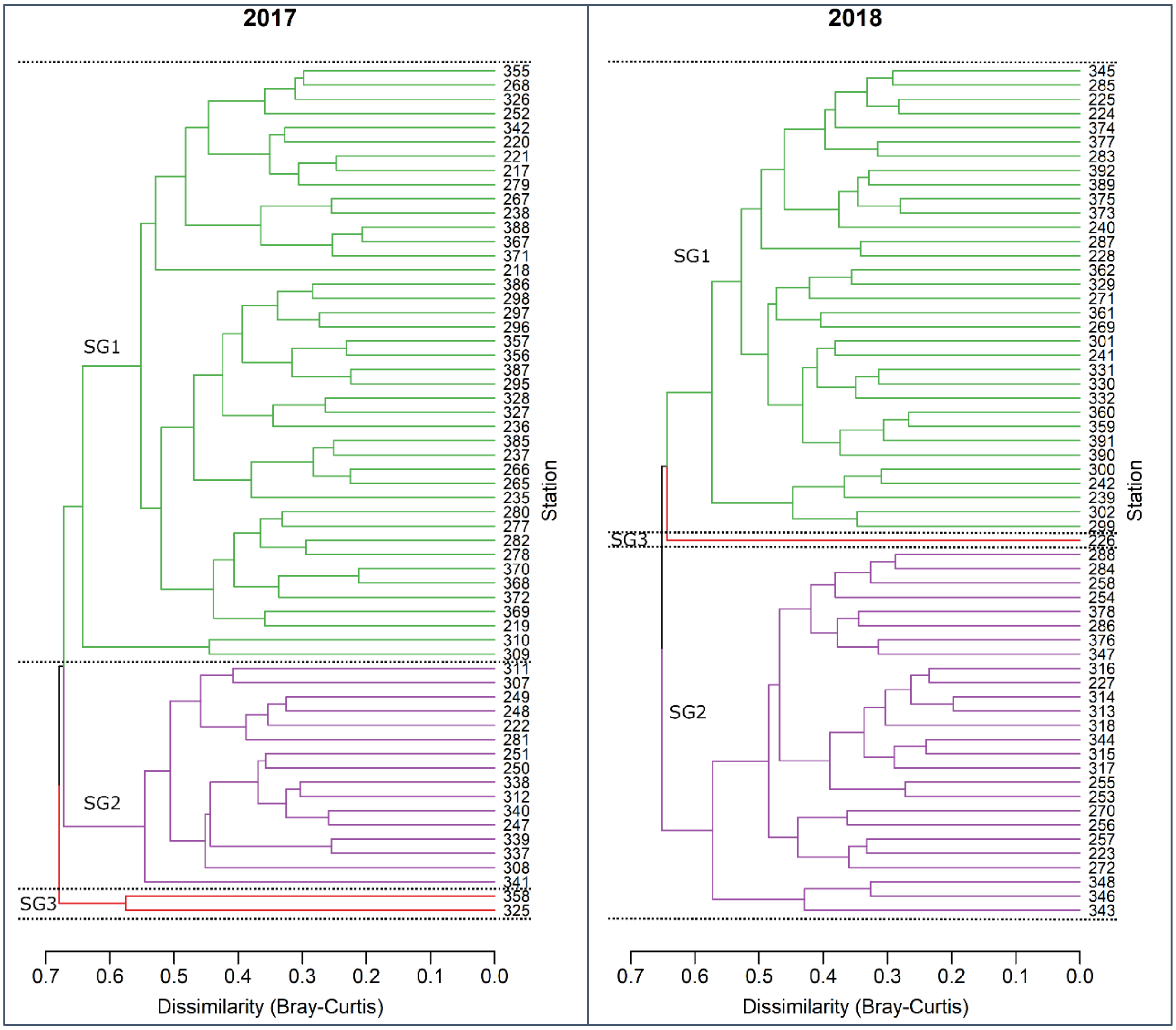

Figure 17. Results of hierarchical clustering of square-root transformed infaunal species abundances (using Bray-Curtis dissimilarities, unweighted pair group method with arithmetic mean) from Mordecai Island in 2017 and 2018. In both years, sites with lower silt+clay content grouped together (purple).

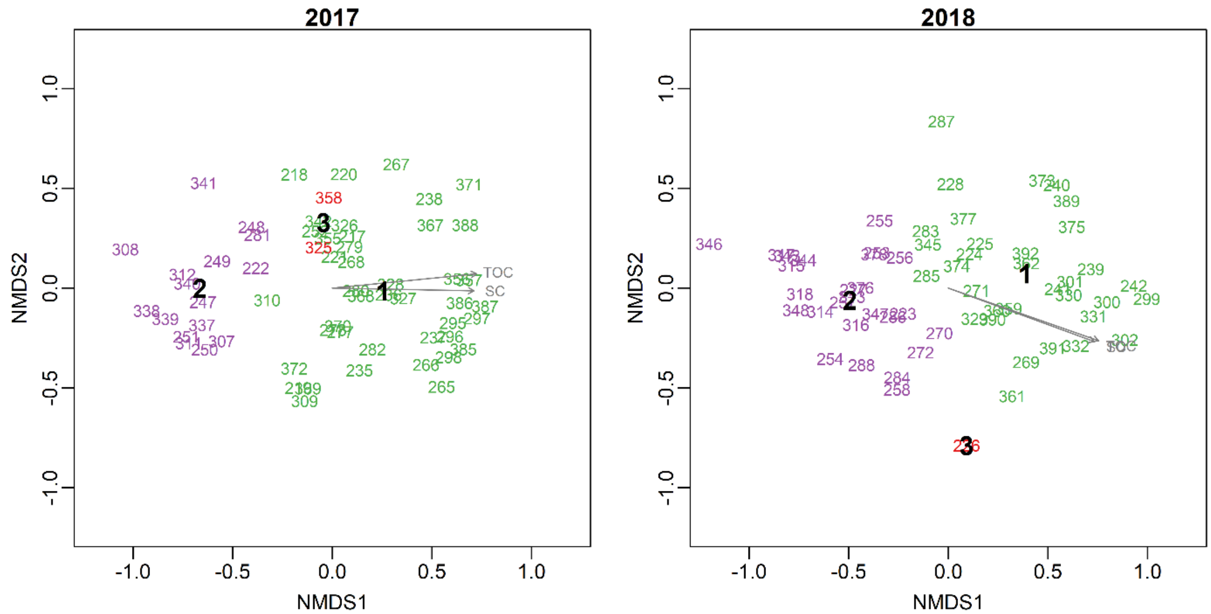

Figure 18. Results of non-metric multidimensional scaling of Bray-Curtis dissimilarities based on square-root transformed infaunal abundance for stations sampled in 2017 and 2018. Group clusters (1 – 3) and station colors correspond to those appearing in Figure 17.

Of the main site groups obtained from hierarchical cluster analysis and illustrated in Figure 17 and Figure 18, Site Group (SG) 1 was dominated by spionid polychaetes. In both years (2017 and 2018), nearly half of SG1 infaunal abundance was represented by Spionidae, which consisted primarily of the highly productive, opportunistic polychaete *Streblospio benedicti*. Known to be tolerant of high organic content (Reish 1979), *S. benedicti* was both dominant and highly abundant among SG1 stations, which tended to have higher percent silt+clay and TOC, as noted previously. The remaining nine dominant taxa in SG1 included other polychaetes, amphipods, and oligochaetes (Table 7). The ten dominant taxa in SG1 made up approximately 90 % of the cumulative abundance, with Spionidae accounting for 44 % and 48 % abundance in 2017 and 2018, respectively.

In contrast, the abundance of taxa in SG2 was more evenly distributed, less dominated by polychaetes, and included taxa characteristic of sandy sediments such as bivalves and amphipods. The ten dominant taxa in SG2 made up 80 % and 86 % cumulative abundance in 2017 and 2018, respectively.

While most of the stations with high diversity tended to have lower percent silt+clay, TOC, and infaunal density (and vice-versa), two stations in 2017 that separated from the others (SG3, composed of stations 325 and 358, shown in red in Figure 18) had high silt+clay and TOC, high species richness and diversity, and low density. The ten dominant taxa from SG3 are listed in Table 7. In 2018, SG3 included only one station (station 226) which stood out as having very low percent silt+clay and TOC accompanied by low values of infaunal species richness, diversity, and density. This site was adjacent to the placement area on the western part of the island (Figure 19) and so may reflect some impact from subsequent dredged sediment placement in the interval between sampling efforts in 2017 and 2018. This location also appeared to experience a slight net elevation gain between 2017 and 2018 (see subsequent Figure 23). Only nine taxonomic groups made up 100 % cumulative abundance at this site (SG3) in 2018, with Spionidae (mainly *S. benedicti*) being the dominant taxon (64.3 % abundance, Table 7). As *S. benedicti* is known to settle in new habitats as a pioneer organism (Garcia-Arberas and Rallo 2004, Levin et al. 1996, Reish 1979), their dominance and low numbers of other taxa would be consistent with recovery after such a disturbance.

Table 7. Abundance of major taxonomic groups (ten dominant taxa) by year and site group (SG). Site groups correspond to those obtained from hierarchical cluster analysis and depicted in Figure 17 and Figure 18. Letters following taxon names indicate taxonomic group: P = polychaete, A = amphipod, O = oligochaete, B = bivalve, N = nemertean.

	2017				2018			
	Taxon	Abun.	%	Cumul. %	Taxon	Abun.	%	Cumul. %
SG1	Spionidae (P)	9881	43.9	43.9	Spionidae (P)	6870	48.0	48.0
	Ampeliscidae (A)	3369	15.0	58.8	Capitellidae (P)	1652	11.5	59.6
	Capitellidae (P)	1533	6.8	65.6	Clitellata (O)	933	6.5	66.1
	Clitellata (O)	1389	6.2	71.8	Ampeliscidae (A)	895	6.3	72.3
	Melitidae (A)	1027	4.6	76.3	Melitidae (A)	751	5.2	77.6
	Orbiniidae (P)	942	4.2	80.5	Lumbrineridae (P)	501	3.5	81.1
	Lysianassidae (A)	802	3.6	84.1	Lysianassidae (A)	384	2.7	83.8
	Lumbrineridae (P)	799	3.5	87.6	Nereididae (P)	352	2.5	86.2
	Nereididae (P)	391	1.7	89.3	Phyllodocidae (P)	329	2.3	88.5
	Goniadidae (P)	345	1.5	90.9	Orbiniidae (P)	289	2.0	90.5
SG2	Ampeliscidae (A)	802	27.8	27.8	Capitellidae (P)	988	19.8	19.8
	Tellinidae (B)	305	10.6	38.4	Tellinidae (B)	746	15.0	34.8
	Paraonidae (P)	287	10.0	48.4	Spionidae (P)	647	13.0	47.8
	Goniadidae (P)	186	6.5	54.8	Ampeliscidae (A)	602	12.1	59.9
	Cirratulidae (P)	141	4.9	59.7	Cirratulidae (P)	521	10.5	70.3
	Veneridae (B)	141	4.9	64.6	Clitellata (O)	231	4.6	74.9
	Aoridae (A)	123	4.3	68.9	Orbiniidae (P)	203	4.1	79.0
	Capitellidae (P)	121	4.2	73.0	Goniadidae (P)	137	2.7	81.8
	Melitidae (A)	103	3.6	76.6	Paraonidae (P)	106	2.1	83.9
	Orbiniidae (P)	95	3.3	79.9	Nemertea (N)	89	1.8	85.7
SG3	Lumbrineridae (P)	21	18.9	18.9	Spionidae (P)	99	64.3	64.3
	Orbiniidae (P)	17	15.3	34.2	Capitellidae (P)	39	25.3	89.6
	Goniadidae (P)	13	11.7	45.9	Phyllodocidae (P)	6	3.9	93.5
	Capitellidae (P)	9	8.1	54.1	Goniadidae (P)	4	2.6	96.1
	Spionidae (P)	9	8.1	62.2	Glyceridae (P)	2	1.3	97.4
	Solecurtidae (B)	8	7.2	69.4	Ampeliscidae (A)	1	0.6	98.1
	Ampeliscidae (A)	4	3.6	73.0	Cirratulidae (P)	1	0.6	98.7
	Lysianassidae (A)	4	3.6	76.6	Oedicerotidae (A)	1	0.6	99.4
	Melitidae (A)	3	2.7	79.3	Syllidae (P)	1	0.6	100.0
	Onuphidae (P)	3	2.7	82.0				

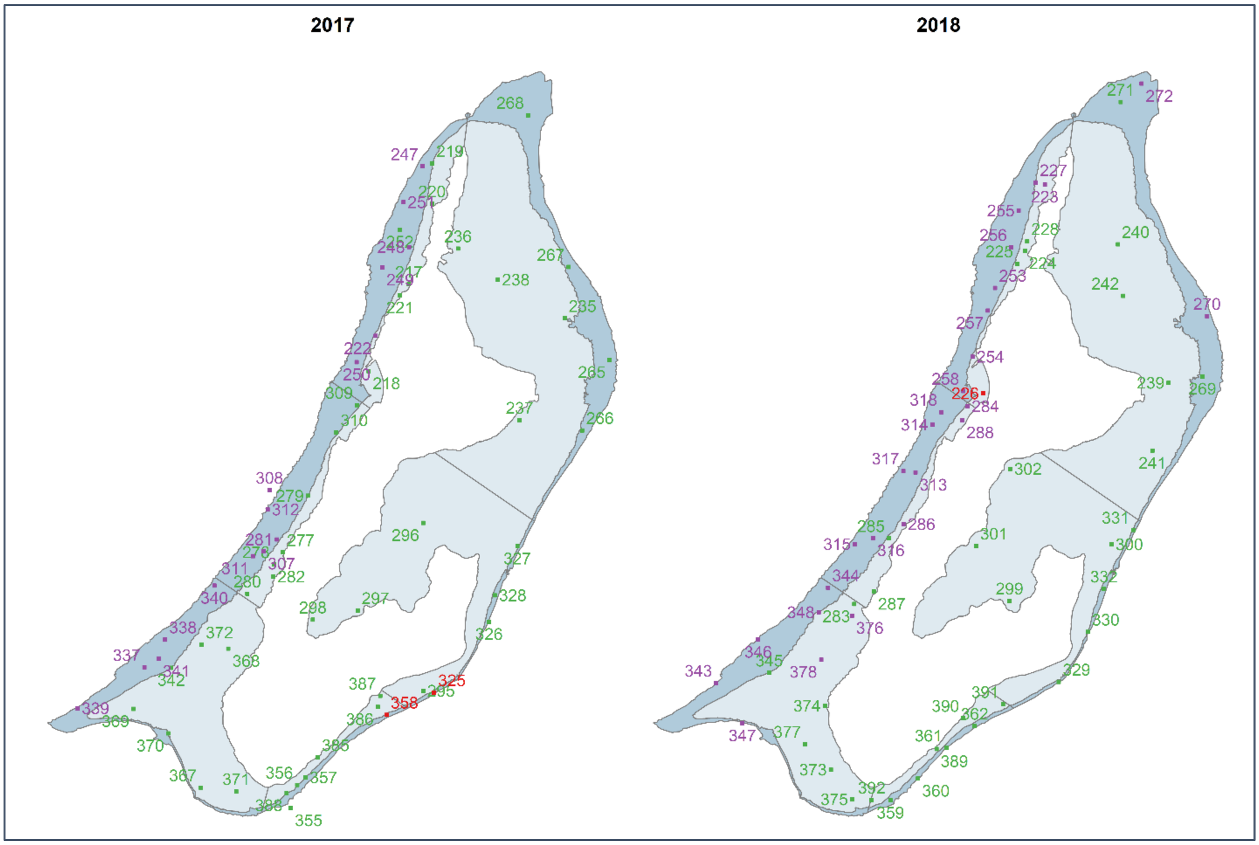

Figure 19. Maps showing the locations and station identifiers of subtidal sediment and infauna collections in 2017 (left) and 2018 (right). Station label colors correspond to those used to identify site groups in Figure 17 and Figure 18.

NMDS analysis of the combined 2017 and 2018 infaunal data reflects the influence of silt+clay noted above, but with no apparent separation by year (Figure 20). Stations sampled in 2018 (represented as open square symbols in Figure 20) are more or less evenly interspersed with stations sampled in 2017 (filled circles). Colors correspond to those used previously in Figure 18.

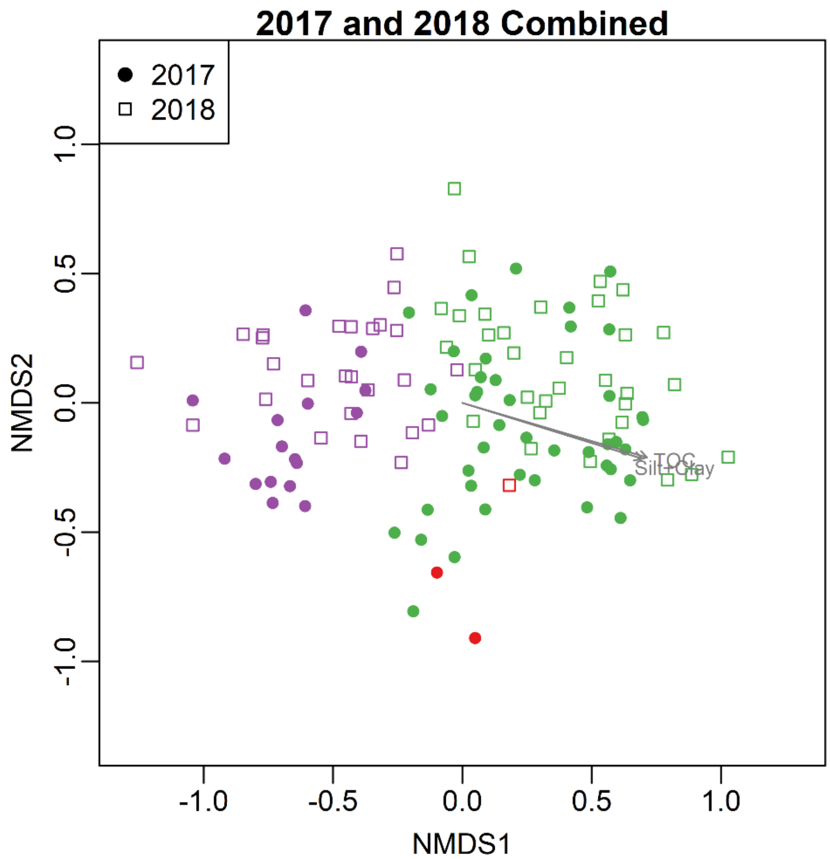

Figure 20. Results of non-metric multi-dimensional scaling of Bray-Curtis dissimilarities based on square-root transformed infaunal abundance performed on the combined 2017 and 2018 dataset. Colors correspond to those used previously in Figure 17, Figure 18, and Figure 19.

3.2.1. Water Quality at Subtidal Sample Stations

Temperatures, salinities, dissolved oxygen concentrations, and depths measured at 120 subtidal stations around Mordecai Island in 2017 and 2018 are summarized in Figure 21. Temperatures ranged from 18 – 24 °C and salinity ranged from 28 – 31 ppt. Dissolved oxygen varied between 3.8 and 12.7 mg/L, well above the range indicative of hypoxia (< 2 mg/L; USEPA 2015). Although stations were pre-selected for one of two targeted depth strata (i.e., shallow sites 0 – 3 ft (0 – 0.9 m) and deep sites 3 – 6 ft (0.9 – 1.8m)), actual depths varied slightly from the targeted ranges, with some shallow sites including depths to 1.4 m, and some deep sites including depths from 0.2 – 2.7 m (0.4 – 3.0 m in 2017). The somewhat lower temperatures and higher salinities observed in 2017 may be a result of unintended differences in sampling conditions, including sampling window (sites were sampled in the second week of October in 2017 vs. the first week of October in 2018), tidal cycle, and depth. While some influence of depth on the distribution of benthic infauna was observed (and discussed previously in section 3.2), no significant associations between infaunal assemblages and temperature, salinity, or DO were found, based on rank correlations with community dissimilarities (function *bioenv*, R *vegan* package) and tests of significance of environmental vectors fitted onto an ordination (NMDS, section 3.2) of the infaunal community matrix (function *envfit*, R *vegan* package).

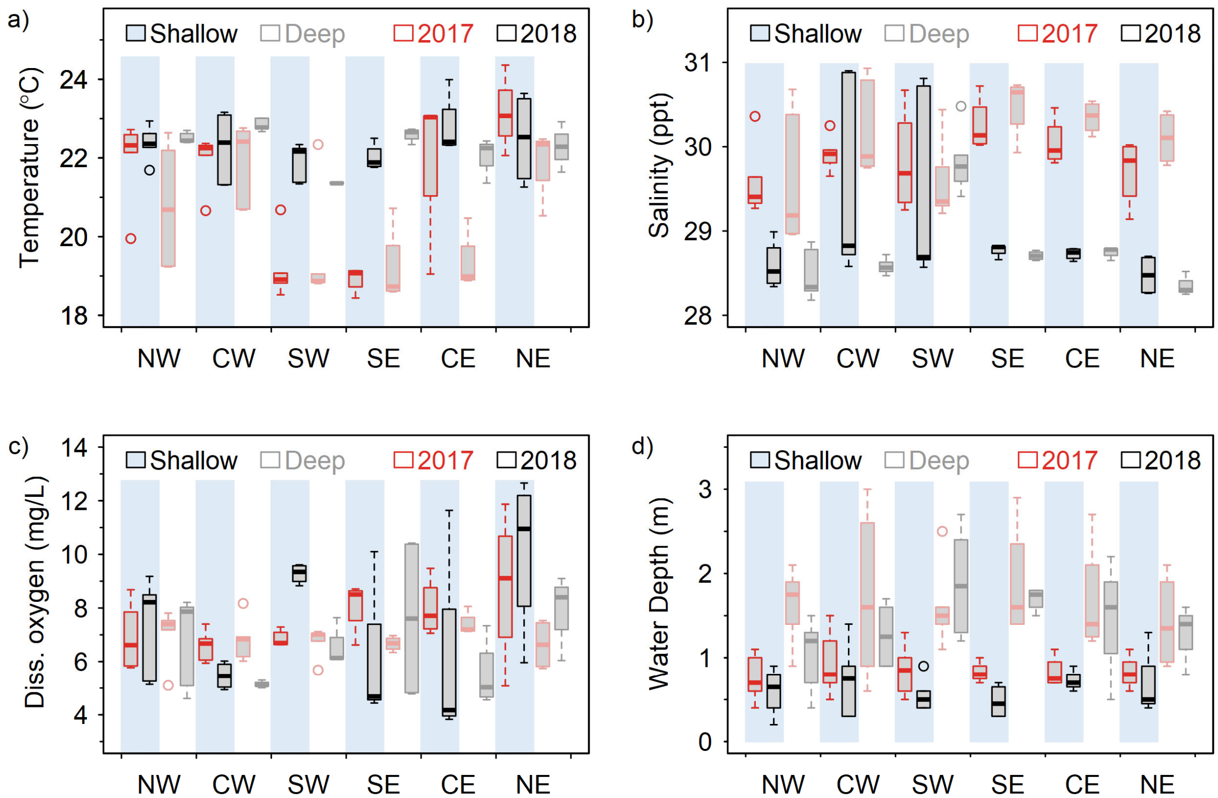

Figure 21. Ranges in water quality parameters measured in 2017 and 2018 at targeted shallow and deep subtidal stations around Mordecai Island. Abbreviations along the x-axis refer to regions of the island (NW=northwest, CW=central west, SW=southwest, SE=southeast, CE=central east, NE=northeast).

3.3. Shoreline Change

Rates of shoreline change were estimated using the End Point Rate (EPR) approach, where EPR is calculated by dividing the net shoreline movement by the total time elapsed between collections. Shoreline change calculated for the entire length of the western shoreline from 1970 to 2017 averaged 0.83 m/yr, which equates to a net loss of 39 m over the 47-year period. In comparison, the rate of shoreline change for the period from 2017 to 2019 (0.39 m/yr) was less than half of the long term average. We suspect that the lower rate of shoreline change from 2017-2019 is partially driven by the shorter time interval. Changes in shoreline position are the result of the cumulative impacts of stressors that range from high frequency, low energy events like daily wind waves, to low frequency, high energy events like hurricanes and nor'easters. As a result, analyses of shoreline change over short time intervals may not be reflective of long term averages because they may not incorporate the full range of conditions experienced. For example, this region of New Jersey was heavily impacted by Hurricane (a.k.a., "superstorm") Sandy in 2012 which came ashore just south of Beach Haven with hurricane-force winds. Any impacts of this storm on Mordecai are incorporated into the long-term average. There were no events of comparable magnitude in the 2017 – 2019 time period. At the same time, the lower average rate of change may be partially driven by real differences in overall rate due to the installation of shoreline erosion control features. The sand-filled geotube structures that flank the island's southwestern shoreline were installed in 2010 to protect a region which, based on historical imagery, had experienced the greatest amount of erosion over the past century. Thus, analyses of shoreline change that include only time periods after 2010 could reasonably be expected to have lower overall rates of change if the geotubes are fulfilling their intended purpose. Additional wave-attenuating structures (oyster castles in varying configurations) were installed to the north of the geotubes (but south of the "natural" area of interest) in 2017 and 2018 which may have further impacted recent rates of shoreline change.

To evaluate the influence of the geotubes on shoreline change, we compared recent rates of change between the geotube and natural areas of interest. These data indicate that from 2017 to 2019, the shoreline of the natural area retreated at a rate of 0.53 m/yr while the rate of retreat of the geotube area was essentially zero (0.01 m/yr). The same comparisons for the time interval 1970 to 2007 (before any shoreline protective features were installed) indicates that the rate of shoreline retreat at the geotube site was lower than that of the natural site (0.9 vs 1.39 m/yr, respectively) but both sites were actively retreating.

Analysis of changes in total areal extent of Mordecai Island indicates that the island decreased from 59 acres in 1970 to 49 acres in 2017, a 17% loss (Figure 22). The 2017 value includes the 6 acres that were gained through dredged material placement; without this, the total loss since 1970 would be 27%. Continued losses of this magnitude could threaten the future stability of Mordecai Island and its role as a wave break for the Beach Haven shoreline.

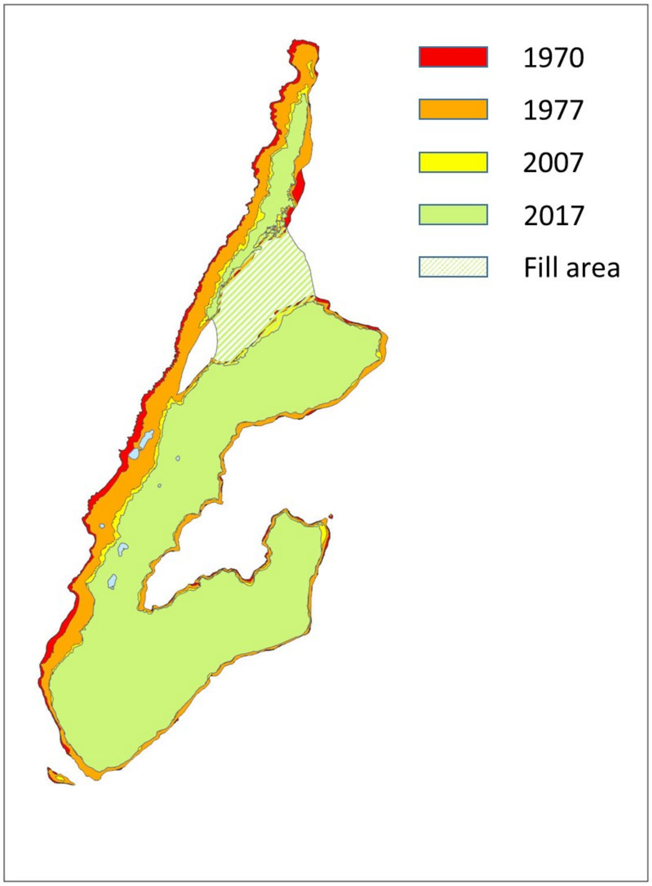

Figure 22. Total areal extent of Mordecai Island by year calculated from aerial imagery-derived shorelines. The hatched green area indicates the sediment placement region.

3.4. Elevation Change

3.4.1. Placement Area

Comparison of the placement region surface elevation profiles between 2017 and 2019 indicates that there has been a net loss of sediments from the northern and western faces of the mound area (Figure 23). The change in profile of the western side of the mound was visually apparent by early 2017 and in response, volunteers from the Mordecai Land Trust and ReClam the Bay installed a series of clam-bag groins parallel to the shore in order to trap the sediments that were being eroded from the mound face (Figure 23). Additional bags were placed in 2018 and 2019 further from the shoreline in response to continued erosion. To date, the effectiveness of these structures has been variable with minimal changes in elevation on the landward side of the low wall of clam bags that runs parallel to the shoreline across the full extent of the placement area (Figures 23 and 24). The four shorter rows of clam bags slightly inland from the wall seem to be trapping sediment at their northernmost extents only. We note that the elevation gain on the top of the mound was due to the additional sediment placement that occurred in winter of 2017.

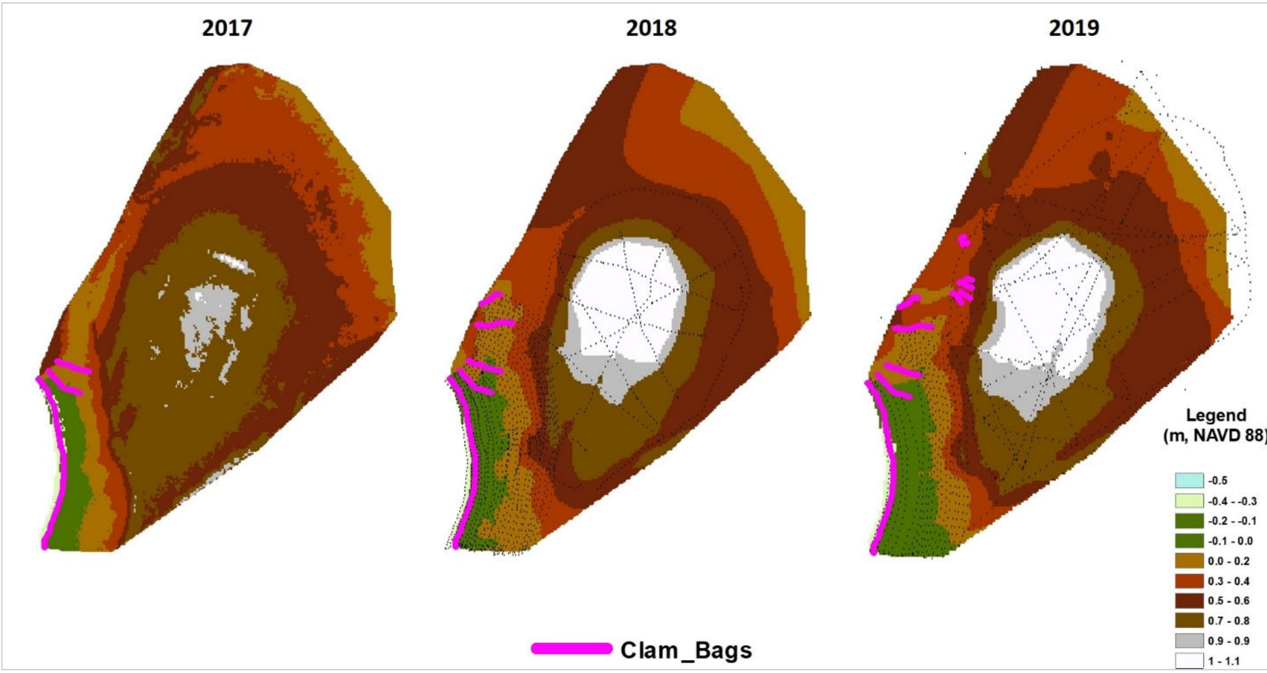

Figure 23. Annual Digital Elevation Models showing changes in sediment surface elevation over time. Black dots represent points where RTK-GPS elevation data were collected as part of surveys conducted with the bike-mounted GPS receiver, described in section 2.2.

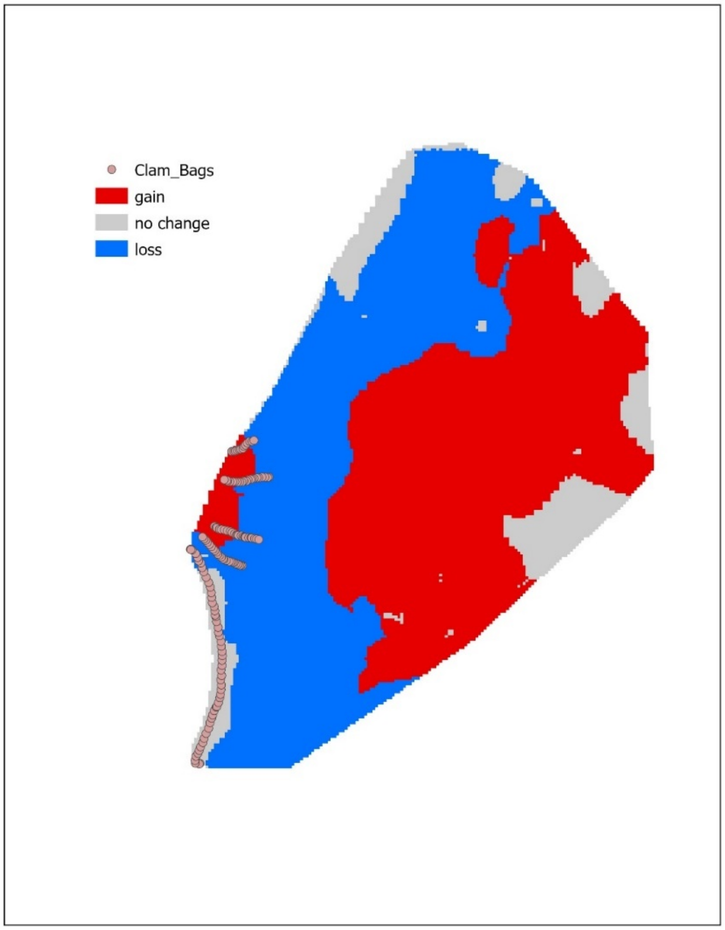

Figure 24. Cut and Fill analysis comparing 2017 and 2019 DEMs of the placement region to illustrate spatial change in elevation over time. Grey areas represent all regions that experienced changes of < 5 cm.

Overall, changes in elevation in the placement region indicate the formation of a channel across the northern boundary of the placement region. At high tide, water floods all the way across the island in this region (Figure 25). Without further sediment addition and/or stabilization (e.g., planting, clam bag placement), this channel has the potential to grow and, ultimately, divide the island back into two distinct lobes.

Figure 25. Sediment placement area near the northern end of Mordecai Island. During higher tides, the boundary between the placement region and northern lobe of the island is flooded all the way across.

3.4.2. Geotube and Natural Areas of Interest

Cut Fill analysis was performed using 2017 and 2019 DEMs from the Geotube and Natural AOIs. Results indicate that the Natural region experienced a net loss of elevation at the shoreline edge (due to edge erosion) and a net increase in elevation of the platform adjacent to the shoreline on the order of 7 cm (Figure 26). Approximations of the amount of sediment lost to shoreline erosion (based on measured amounts of edge retreat and using an estimate of 0.5 m for depth of marsh sediment) and the amount required to support the elevation increases measured on the adjacent platform suggest that the two are nearly in balance. The marsh adjacent to the geotube structure where the erosion rate is substantially lower also experienced a net increase in elevation although at a lower rate (4 cm increase on average). Essentially, edge erosion is likely providing at least some of the sediment that is fueling elevation growth in the nearshore marsh platform, particularly along the rapidly eroding stretches. Correlation between rates of shoreline erosion and elevation growth of the adjacent marsh platform has been demonstrated previously in other marsh systems (Currin et al 2015). Regardless of the source of sediments, both the natural and geotube AOIs are increasing in elevation at a rate that is an order of magnitude greater than the local long-term rate of relative sea level rise (station #8534720, Atlantic City, NJ; 4.12 ± 0.15 mm yr^{-1}; 1911 – 2019).

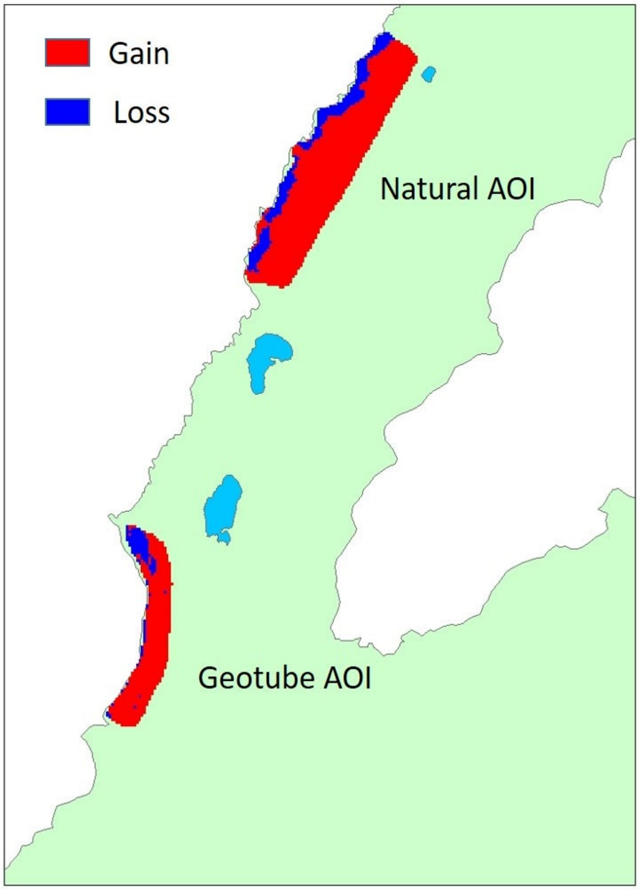

Figure 26. Elevation change (2017 to 2019) in Natural and Geotube AOIs. Losses represent edge erosion while gains represent elevation growth of the adjacent marsh platform.

3.5. Submerged Aquatic Vegetation and Benthic Macroalgae

Out of 60 benthic stations sampled in 2018, SAV was documented at seven stations, and always in combination with macroalgae (Figure 27). The dominant seagrass in this region is *Zostera marina* (Kennish et al. 2008) and that was the seagrass that appeared to be present in all images. Algae (almost always a mixed community of red filamentous and green (*Ulva* sp.)) was detected at 18 stations. There were 22 stations where vegetative cover was present but the type of vegetation could not be determined conclusively due to poor image quality resulting from high turbidity due to windy conditions during sampling. The seven stations where seagrass was definitively identified were all located on the western side of the island. Visual evidence (on days with high water clarity) suggested that many of the sampling stations on the eastern side with “undetermined” vegetative cover were occupied by algae. This is corroborated by observations from the benthic grab samples as we often pulled up algae at these stations in the process of attempting to collect bottom sediments.

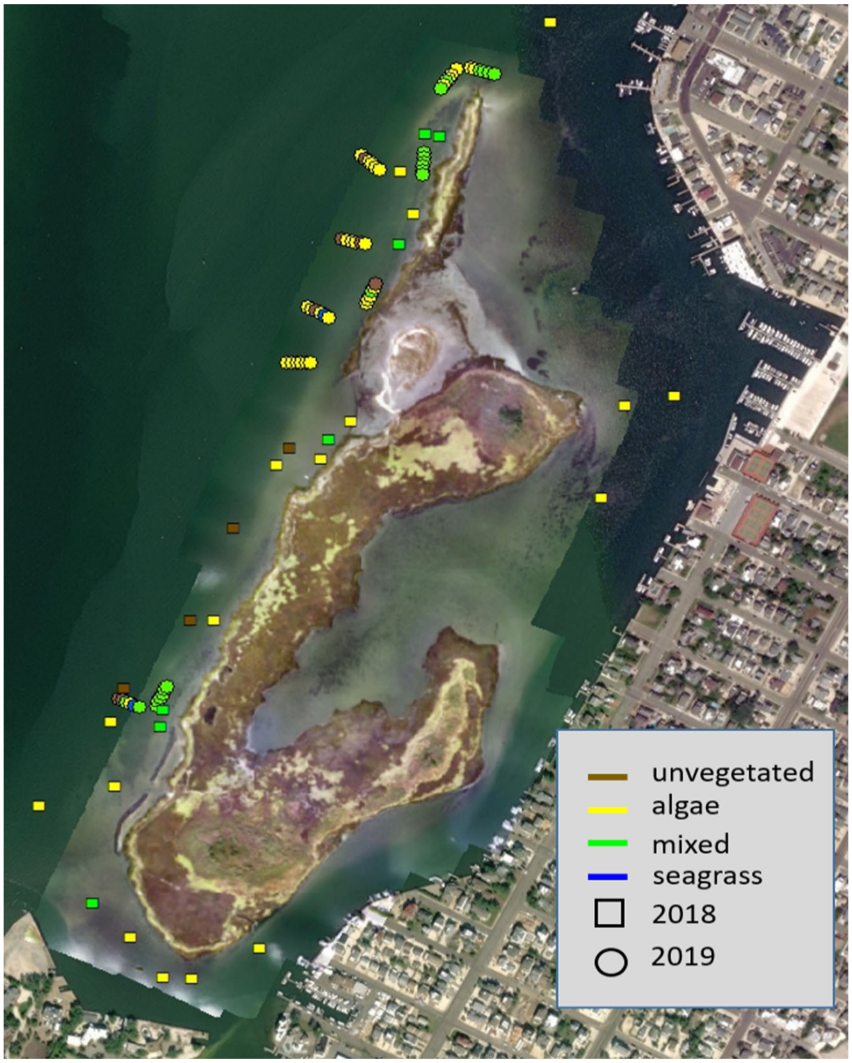

Figure 27. Occurrence of seagrass and macroalgae at subtidal sampling stations in 2018 and 2019. Stations where both seagrass and algae were documented are labeled as "mixed".

In 2019, underwater image collection was concentrated in the nearshore waters of the Northwest region as this area was identified by partners as being of higher priority due to planned future modifications to that portion of the shoreline. In this region, SAV was present in mixed assemblages with macroalgae in the majority of nearshore sample locations (Figure 27), though macroalgae was generally more abundant than SAV. Total percent cover of both algae and SAV dropped off dramatically with distance from shore. The high-resolution UAS-based imagery collected in the northwest region was valuable for identifying areas with dense vegetative cover although it is not possible to distinguish between seagrass and macroalgae from this product (Figure 28). As of 2019, there was a dense strip of macroalgae growing adjacent to the clam bags that were installed across the western perimeter of the placement region. Outside of this belt of macroalgae, there was only sparse vegetative cover in the region offshore of the placed sediments (Figure 28, position e).

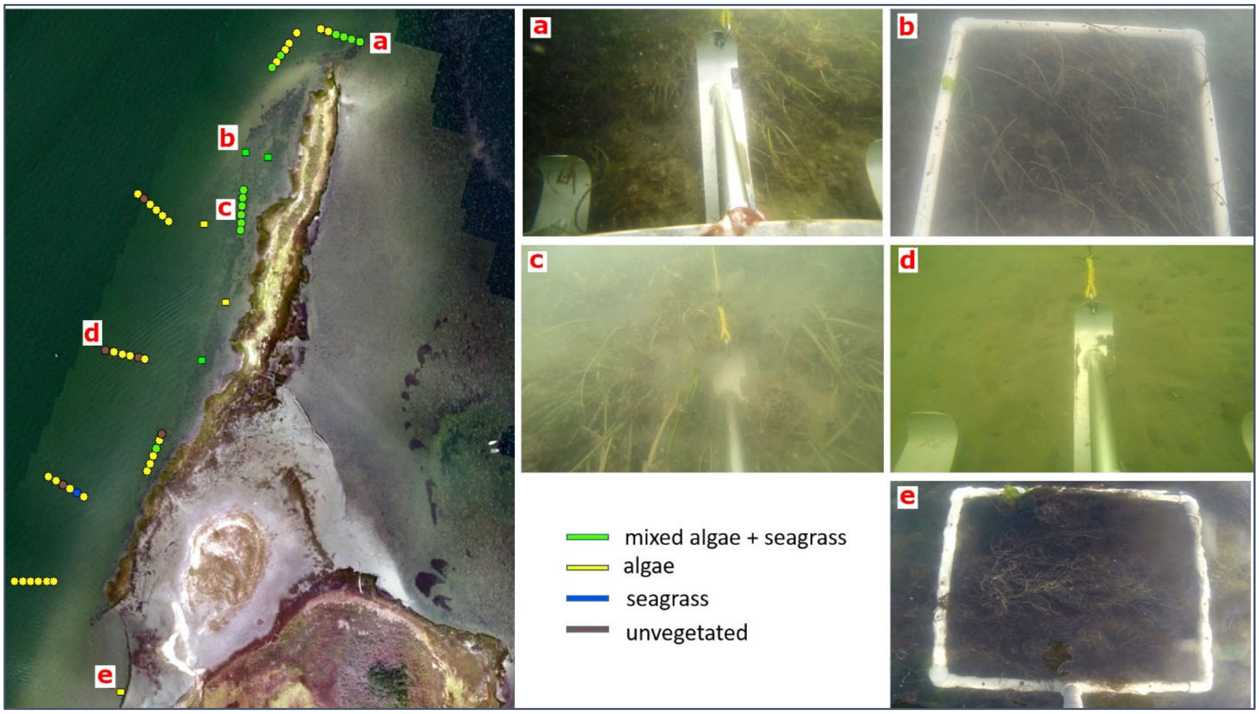

Figure 28. Locations of underwater imagery used to ground truth aerial imagery and estimate percent cover of SAV and benthic macroalgae (2018 - point locations, indicated by filled squares; 2019 - transects, indicated by filled circles).

4. Summary and Conclusions

The beneficial use of dredged sediments to create and enhance NNBF can contribute to the long-term resilience of coastal regions by keeping valuable sediment in its system of origin (Bridges et al. 2015). Through strategic placement in critically degraded habitats like Mordecai Island, beneficial reuse projects have the potential to increase the ecosystem services and protective value of the habitats in which they are placed. On a nationwide scale, there are likely to be numerous opportunities to match navigation dredging with habitat restoration for enhancement of coastal resilience. Despite the opportunities, these strategies have yet to be embraced widely by practitioners and regulators due to uncertainties about the long-term performance of these habitats relative to more traditional engineered solutions (Morris et al. 2018; Moeller 2019). The data provided here partially address that knowledge gap by providing the baseline against which to evaluate the success of beneficial use at Mordecai Island.

4.1. Trajectory of Marsh Development

Following dredge spoil placement, vegetation in the newly created intertidal regions of Mordecai Island has been slow to develop and many large patches remained unvegetated as of fall 2019. The initial planting was done in spring of 2016 and by the time of the first data collection in the fall of 2017, the dead stems of these plantings were still apparent. It is not clear what led to the poor performance of these plantings as the substrate/porewater conditions and elevation appear to be suitable for growth. Extensive grazing by geese (observed by local site managers) and low nutrient availability in the sandy sediments may have contributed to the poor vegetative success. Still, some of the plantings survived and as of 2019, the unvegetated areas of the placement area were beginning to be colonized by lateral growth of *S. alterniflora* from areas where plantings survived as well as from the natural marsh surrounding the placement area. We anticipate that without further intervention and in the absence of further perturbation, this trend will continue and that with time, a more species-rich vegetative community, similar to that of the natural marsh, will develop. Additional plantings would likely speed up the process of recolonization.

The invasive plant *P. australis*, which is known to be highly effective at colonizing disturbed, well-drained sediments (Bart and Hartman, 2000), was already present in dense stands on the island at elevations > 0.6 m NAVD88 prior to the restoration. The central area of the placement region was originally raised to an elevation of 1 m and then subsequently raised to an elevation of 1.2 m NAVD88 with the second deposition in December 2017. This high elevation mound, intended to remain devoid of vegetation to provide nesting habitat for birds, was rapidly colonized with *P. australis* which forms monotypic stands characterized by an extensive system of underground rhizomes which contribute to sediment stability and elevation growth. Thus, while *P. australis* colonization has decreased the value of the mound for nesting habitat, it may ultimately contribute to the stability of the placed sediments. Efforts to fully eradicate *P. australis* in other wetlands have met with limited success (Quirion et al. 2018); however, with continued application of mechanical (e.g., burning or cutting) or chemical (e.g., glyphosate) control techniques, it may be possible to maintain suitable nesting habitat in the placement region.

4.2. Impacts to Subtidal Benthic Habitats

4.2.1. Benthic Infauna

This study was unable to detect any significant impacts to benthic infaunal assemblages in the subtidal areas around Mordecai Island that could be attributed directly to dredged sediment placement activities. The strongest determinants of benthic metrics (infaunal abundance, richness, and diversity) appeared to be those related to sediment grain size (percent silt+clay vs. sand) and organic content (TOC), rather than proximity to sediment placement areas. In general, species richness and diversity tended to be higher (and densities lower) at stations along the western side of Mordecai Island, which is a higher-energy environment exposed to wave action from the bay and characterized by sandy sediments with low silt+clay content. Some exceptions (i.e., low species richness, diversity, and density) included stations adjacent to the western portion of the placement area, but similar values of benthic metrics were also observed farther south along the western side of Mordecai at stations unlikely to have experienced any impacts from sediment placement activities. Complicating the ability to measure benefits to or impacts on benthic infaunal communities from sediment additions is the lack of pre-placement data. Although the data reported here provide an important baseline for monitoring future changes to infaunal assemblages, it is not possible to assess how post-placement conditions differ from those prior to project initiation since no samples were collected before the start of the project. Hence, it is strongly suggested that similar projects in the future having a benthic component also include a pre-placement sampling component to allow comparison of conditions before and after sediment placement.

4.2.2. Submerged Aquatic Vegetation

The vegetative community in the subtidal regions surrounding Mordecai is dominated by macroalgae. Seagrass occurred in small but dense stands (usually mixed with algae) on the western side of the island and always in close proximity to the shoreline. We did not identify any SAV in the subtidal zone adjacent to the western boundary of the placement area. It is not clear if SAV previously occurred in that region but was disturbed by placement activities, or if that region of the western shoreline was already devoid of seagrass before placement activities began. We did not detect any seagrass on the eastern side of the island but did find dense assemblages of macroalgae. The data collected to date do not provide insight into the potential impacts of placement activities on SAV occurrence or abundance.

4.3. Implications for Long-term Resilience of Mordecai Island

Changes in the elevation profile of the placement region between 2017 and 2019 indicate a loss of sediments from the unvegetated region to the north of the mound. Continued loss of sediments from this region seems likely to result in the formation of a channel and, ultimately, lead to the island being separated once again into two lobes. Regrading of the sediments in this region (with or without additional material placement) followed by dense planting is advisable to retard further channel formation. As of late 2019, the surface elevations within this region were within the optimal range for growth of *S. alterniflora*. At a minimum, dense planting within this area should be considered as the presence of vegetation will not only serve to defend against further loss of sediments, but should help to trap new sediments and promote stability. Clam bags are helping to slow the loss of material from the system but their impact is spatially limited.

Analyses of historical shoreline positions indicate that between 1970 and 2019, the western shoreline of Mordecai retreated at an average rate of 0.83 meters per year leading to a cumulative loss of roughly 15 acres of land. This rate is in agreement with previously-reported erosion rates in the Barnegat Bay - Little Egg Harbor System (Leonardi et al. 2016) where Mordecai Island is located. This previous effort documented change across roughly 100 km of shoreline from 1930 to 2007 and from 2007 to 2013 and showed that the average rate of shoreline change was consistent over the two time intervals. Shoreline erosion is a major source of the sediments that end up clogging navigation channels and leading to the need for maintenance dredging; it is also a cause of habitat loss for marsh islands like Mordecai. Beneficial use of sediments to restore these habitats can be a part of the solution. Continued data collection to further document the rate of ecosystem service provision and protective capacity of demonstration sites like Mordecai Island will play a pivotal role in determining the efficacy and optimal use of such applications.

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