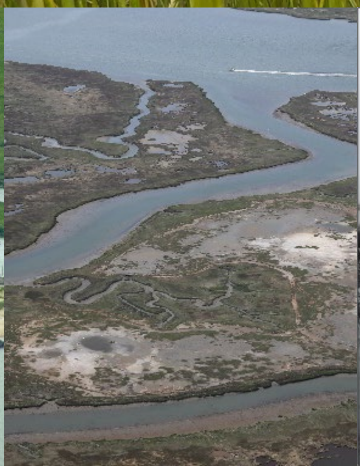
Beneficial Use of Dredged Material to Enhance Salt Marsh Habitat in New Jersey

Monitoring and Project Assessment

January 2023





Acknowledgments:
This project was supported and funded by the National Fish and Wildlife Foundation through the Hurricane Sandy Coastal Resiliency Competitive Grant Program (Grant #43095), The U.S. Environmental Protection Agency Wetland Program Development Grant (CD-96273100), and a U.S. Fish and Wildlife Wildlife Restoration Grant (Grant #F17AF00813). The project was sponsored by the New Jersey Department of Environmental Protection (NJDEP), with support from the Office of Dredging and Sediment Technology, the Division of Science, the Bureau of GIS, and the Division of Fish and Wildlife. Additional support was also provided by project managers, research scientists, and engineers from the NJDEP, the New Jersey Department of Transportation – Office of Maritime Resources, the U.S. Army Corps of Engineers (USACE) - Philadelphia District, the USACE Engineer Research and Development Center, GreenVest LLC, Princeton Hydro LLC, The Wetlands Institute, Rutgers University, Stockton University, and The Nature Conservancy in New Jersey.

Recommended citation*: New Jersey Department of Environmental Protection and The Nature Conservancy (2023). Beneficial use of dredged material to Enhance Salt Marsh Habitat in New Jersey: Monitoring and Project Assessment. (Eds. M. Yepsen, B. Wilburn, J. Woollard). Trenton, NJ. 122 pp.*

Title page photo credits: Metthea Yepsen (NJDEP Division of Science and Research), Steven Jacobus (NJDEP Bureau of GIS), and ACE Aerial Images and Video Production

Abstract:
This report summarizes monitoring conducted at three pilot beneficial use of dredged material to enhance salt marsh projects in New Jersey. Constructed between August 2014 and April 2017, these projects tested sediment addition techniques that included thin-layer placement (TLP) of dredged material on the platform of vegetated, stressed marshes (Ring Island, Avalon, and Fortescue) and the filling of degraded and expanding pool-panne complexes with dredged material on the surrounding stressed marsh platform (Avalon). The objectives for the three marsh pilot projects were (1) to increase and maintain the optimal tidal elevation (hydroperiod) for native salt marsh species, (2) to increase the cover and health of native salt marsh vegetation, and (3) to return all other metrics to baseline (*i.e.*, pre-implementation) conditions (unless they were expected to change due to habitat conversion). Topographic surveys indicated that 1) on average sites reached target elevations, but the placement was uneven, 2) all sites initially gained elevation, but it was challenging to measure small elevation changes, 3) the higher the final elevation, the slower vegetation grew back, and 4) sites gained resilience against 10- to 27-years' worth of sea-level rise. As of 2021, none of the salt marsh sites had increased plant cover from baseline conditions or established the targeted *Spartina patens* habitat. However, several sites matched control site conditions, and much was learned about how to increase the rate of plant recovery. Soil makeup, benthic infauna communities, and epifaunal macroinvertebrates did not return to baseline conditions by 2021, but water chemistry returned to control conditions. Nekton and avian use were variable and results were dependent on changes to vegetation and elevation. These findings suggest that both thin- and thick-layers of sediment addition to existing tidal marshes led to large initial changes in the habitat, from which the ecosystems rebounded/are rebounding at different rates.

Editors:
Metthea Yepsen, NJ Department of Environmental Protection and formerly The Nature Conservancy
Brittany Wilburn, NJ Department of Environmental Protection and Drexel University
Jaclyn Woollard, formerly at the NJ Department of Environmental Protection and The Nature Conservancy

Contributors to this report include:
Allison Anholt, Formerly with The Wetlands Institute
Jessie Buckner, Formerly with The Nature Conservancy
Brian Cramer, GreenVest, LLC
David DuMont, The NJ Department of Environmental Protection
Doug Faircloth, Formerly with The Nature Conservancy
Alex Ferencz, Stockton University
Lisa Ferguson, The Wetlands Institute
Robert George, Princeton Hydro
Dave Golden, NJ Department of Environmental Protection
Steven Jacobus, The NJ Department of Environmental Protection
Jaqueline Jahn, Formerly with GreenVest, LLC and The Nature Conservancy
Moses Katkowski, Formerly with The Nature Conservancy
Evan Kwityn, Princeton Hydro
Lee Lippincott, The NJ Department of Environmental Protection
Laura Moritzen, Formerly with The Nature Conservancy
Lori Lester, The NJ Department of Environmental Protection
Elizabeth Muntean, Formerly with The Nature Conservancy
Mary Paist-Goldman, Formerly with Princeton Hydro
Nick Procopio, The NJ Department of Environmental Protection
Michael Rehman, Princeton Hydro
William Smith, The NJ Department of Environmental Protection
John A. (Jack) Szczepanski, Princeton Hydro
Gary Taghon, Rutgers University
Lenore Tedesco, The Wetlands Institute
Rob Tunstead, USDA Natural Resources Conservation Service
Brittany Wilburn, NJ Department of Environmental Protection and Drexel University
Jaclyn Woollard, Formerly with the NJ Department of Environmental Protection and The Nature Conservancy
Adrianna Zito-Livingston, The Nature Conservancy
Metthea Yepsen, The NJ Department of Environmental Protection

For additional information contact:
Metthea Yepsen at metthea.yepsen@dep.nj.gov

Executive Summary

Due to the growing evidence that tidal wetlands of the Northeast United States are not keeping pace with sea-level rise (Cahoon 2015, Haaf et al. 2022, Hartig et al. 2002, MERI 2015, US EPA 2019), there is an effort to find mechanisms to enhance their resilience and prolong their existence. This report summarizes the monitoring conducted at three pilot resilience projects in New Jersey where sediments taken from channel beds as part of navigational dredging projects were pumped onto the marsh to increase the salt marshes' elevations. The objectives for the three marsh pilot projects were (1) to increase and maintain the optimal tidal elevation (hydroperiod) for native salt marsh species, (2) to increase the cover and health of native salt marsh vegetation, and (3) to return all other metrics to baseline (*i.e.*, pre-implementation) conditions (unless they were expected to change due to habitat conversion).

Monitoring was conducted at the salt marsh enhancement projects between 2014 and 2020. Where possible, monitoring followed a Before-After-Control-Impact (BACI) design. Metrics were selected to see how closely the projects were built as designed to determine whether the project objectives listed above were met and to inform adaptive management. More than 50 metrics were collected by nine organizations over seven years in three sites and associated control sites. Using these data and associated reports, we can answer the following questions:

1. Were projects built as designed?

All marsh enhancement sites were smaller than planned, especially Fortescue. Ring Island was 0.89 acres of the targeted 1.0-1.4 acres. Avalon was 45 of the planned 51 acres, and Fortescue was 6.5 of the targeted 23 acres. The reasons for this are discussed in the 2021 Beneficial Use of Dredged Material to Enhance Salt Marsh Habitat in New Jersey: Project Summary and Lessons Learned report.[1]

The dredged material was generally placed at the Target Dredged Material Placement Elevations in around 50% of each marsh enhancement site. The remainder of the sites was generally higher than planned at Ring Island, lower than planned at Fortescue, and both above and below at Avalon. By 2019, Avalon had a fairly even mix of areas that were above, below, and within the site's Target Ecological Elevations. Elevation gains at Fortescue were no longer measurably different from baseline conditions by 2019 (both because the reported change was too small to be measured by a real-time kinematic global positioning system (RTK-GPS) and because of questionable gains and losses in areas of the site that did not receive sediment).

2. Did the projects gain elevation? If so, how much?

In 2019, Ring Island had maintained the initial elevation gain (a median increase of 0.55 feet), Avalon had maintained some of the initial elevation gain (a median increase of 0.34 feet), and an elevation

[1] https://hdl.handle.net/10929/97940

change could no longer be reliably measured at Fortescue (a median measured increase of 0.2 feet).

3. How many extra years of elevation capital did the sites gain relative to sea-level rise?

The current short-term rate of sea-level rise in New Jersey is 6 mm/year (Haaf et al. 2016), which is equivalent to 0.02 feet. Given the median elevation gains at the sites measured in 2019, Ring Island, Avalon, and Fortescue gained 27.5, 16.5, and 10 years, respectively.

4. Were the elevations at optimal tidal elevations (hydroperiod) for native salt marsh species?

All projects stated the desire to increase marsh elevation to high marsh, targeting primarily *Spartina patens*. Although Avalon and Fortescue had large areas in the target tidal elevation (between Mean High Water (MHW) and Mean Higher High Water (MHHW) levels) for *Spartina patens* and the species was planted at the sites, it did not become the dominant species.

At Ring Island, where elevation targets were not used, we saw that the addition of a median of 0.55 feet of elevation moved the site from low marsh to above high marsh, which was perhaps too high. Raising the elevation above MHHW may explain why the site has been slower than Fortescue and Avalon to revegetate. At Avalon, three out of the five placement areas had Target Ecological Elevations above MHHW, and anecdotal evidence from site visits has suggested that persistent bare areas tend to correlate with the lowest or highest elevations. Fortescue targets were consistent with the upper end of the high marsh range.

There are inherent challenges in targeting high marsh elevations. The range of elevations that fall into the high marsh tidal datums (approximately 0.5 feet) is much smaller than the range of elevations that fall into the low marsh tidal datums (generally 2-3 feet in marshes in SE New Jersey), creating some inherent challenges for using TLP techniques to reach high marsh elevations because the room for error is much smaller.

5. Was there an increase in the cover and health of native salt marsh vegetation?

The short answer is no, vegetation loss was widespread where sediment was placed at all three sites and the vegetation cover not only did not surpass baseline conditions, but also was still below the baseline six to seven years after construction. However, the sites quickly recovered native species richness and maintained native salt marsh plant communities. In addition, plant cover in most control sites dropped over time resulting in no significant difference in cover with placement sites.

6. Why did vegetation recover quickly in some areas and not in others?

The differences in recovery between the three sites can be attributed to a variety of potential factors, including differences in how the projects were constructed and the environmental characteristics of the sites. We found three main driving forces affecting revegetation rates: 1) for the first few years, vegetation recovery was higher in areas with thinner placement; 2) vegetation was denser in areas near existing vegetation, suggesting that higher edge-to-placement ratios may speed up the recovery rate;

and 3) while final elevation is important in long-term plant recovery (as evidenced at Ring Island), placement depth was not (as evidenced by vegetation growth in Avalon's pool plots).

7. Did all other metrics return to baseline conditions?

Soil makeup, benthic infauna communities, and epifaunal macroinvertebrates had not returned to baseline conditions.

- While control site soils were 90% silt and clay, placed sediments at Ring Island and Fortescue were predominately sand. Placed sediments had much lower concentrations of total phosphorus, total nitrogen, total sulfur, and organic matter than control sites.
- There was an immediate decline in the abundance of benthic infauna across all three sites. Ring Island showed minimal recovery of any taxa four years post-placement and the abundance was similar between placement and control sites at Avalon and Fortescue in 2019. The proportion of benthic infauna taxa in the samples remained different in placement sites through 2019.
- The density of *Melampus bidentatus* (salt marsh snails) remained lower in Avalon and Fortescue placement areas when compared with baseline and increased at Ring Island. There was no measured change in *Guekensia demissa* (ribbed mussels), likely because of the small number of plots where mussels were found. *Uca* sp. (fiddler crabs) burrows decreased at Fortescue and increased at Avalon placement areas when compared with the baseline.

Water chemistry was generally similar in placement and control areas; however, surface water chemistry monitoring documented periods of high pH and salinity in contained areas in the year following sediment placement at Avalon. Tidal flushing was and important moderator of more extreme surface water chemistry conditions. Ground water pH at Avalon slightly increased at placement sites from initial measurements in 2017 to 2019. There was a larger range in groundwater pH and surface water salinity measurements at Avalon placement sites than in control areas and water chemistry evolution was documented as the site aged.

The placement of dredged material on the marsh surface initially made acute changes to the habitat available to birds using each project site. Shorebirds increased with the creation of open, sparsely vegetated areas, and marsh-dependent species decreased. Marsh species began to rebound as vegetation recovered. Reproductive success at Ring Island was comparable to control areas in the marsh. The marsh enhancement areas of Ring Island supported fewer species and fewer nests compared to the Elevated Nesting Habitat and control areas, but the number of species nesting in these areas increased over the five-year monitoring period.

Nekton was variable based on changes to vegetation and elevation. Decreased fish, decapod, and total nekton density were observed in placement areas at Fortescue and Ring Island during the first and second post-restoration years. This was expected because increasing elevation and removing pools were expected to decrease nekton access and habitat.

Key takeaways:

1. A dredged sediment slurry can be pumped over large areas of salt marsh and generally reach target elevations, but a high level of variability in thickness and elevation is to be expected without the use of grading equipment or new technology.
2. Even a very thin layer of sediment can kill existing marsh vegetation.
3. The thickness of the sediment is less important than the final elevation's effect on flooding in the rate of plant regrowth.
4. There are tradeoffs between the rate of vegetation regrowth and elevation gain.
5. Plants tend to regrow from the edges of placement, suggesting that a higher edge to the interior area of placement ratio may increase the rate of revegetation.
6. *Salicornia* sp. (commonly called sea beans, glasswort, or pickleweed) were the first plants to recolonize the placement areas but were subsequently replaced by *Spartina alterniflora* (smooth cordgrass) and *Distichlis spicata* (saltgrass) as a result of planting and natural recruitment.
7. It is hypothesized that changes in animal species are a result of changes in the habitat. Bare areas with little organic matter in the soil are not hospitable to typical marsh birds and benthic infauna. They instead drew fiddler crabs, shorebirds, and opportunistic benthic infauna that had a higher tolerance for disturbed sites.

Next Steps: The sediment addition and control sites will continue to be monitored as part of the New Jersey Tidal Wetlands Monitoring Network and publications based on the monitoring are being prepared for peer-reviewed publications.

Table of Contents

Glossary

The following terms have been defined according to how they have been specifically used in this document and in the context of the marsh enhancement projects discussed herein. These definitions may differ from those used by others for the same/similar terms.

Adaptive management—The process of continually, iteratively reviewing data and information to develop, design, construct, monitor, and manage a project, in order to meet the project objectives.

Beneficial use of dredged material—The placement of dredged material to enhance, create, or restore a variety of habitats, as opposed to the usual practice of disposing of it in a confined disposal facility or other practices that remove sediment from the system.

Biological benchmarks—The optimum tidal elevation range of the plant species of interest. These benchmarks are used (at least, in part) to design a habitat (e.g., marsh) enhancement and to establish Target Ecological Elevations for enhancement projects.

Coastal resiliency—The ability of coastal communities and natural habitats to withstand and recover from the impacts of storms, flooding, accelerating sea-level rise, and other natural hazards.

Dredging team—Personnel whose primary responsibility is to ensure the dredging success of the projects.

Ecological resiliency—The ability of an ecosystem (e.g., a salt marsh) to resist, respond to, and recover from a disturbance (e.g., an increase in the rate of sea-level rise).

Elevated Nesting Habitat (ENH)—Colonial shorebird nesting habitat created by placing sandy dredged material in a mound on the marsh platform to increase the elevation of the land and create sparsely vegetated sandy habitat. The ENH is high enough to protect the nests from flooding during high tides and storms.

Enhancement—The improvement of a wetland's ability to support natural aquatic life, through substantial alterations to the soils, vegetation, and/or hydrology, as defined by New Jersey Regulations. Also called "restoration" in other parts of the country.

Marsh team—Personnel whose primary responsibility is to ensure the ecological success of the projects.

Mean High Water (MHW)—The average of all the daily tidal high water heights observed over a period of several years.

Mean Higher High Water (MHHW)—The average height of the highest tide recorded at a tide station each day during the recording period.

New Jersey Intracoastal Waterway (NJIWW)—A system of navigation channels maintained by the U.S. Army Corps of Engineers that extends along the New Jersey Coast from the Atlantic Ocean at Manasquan Inlet to Delaware Bay, about 3 miles north of Cape May Point.

Placement areas—Specific areas on the marsh selected for enhancement that received sediment dredged from a nearby navigation channel or marina. Also called "marsh enhancement areas" or "dredged material placement areas."

Stressed marsh—Marshes in need of enhancement (not a regulatory term). Characteristics of stressed marshes include eroded edges, expanded and degraded pools with undercut banks, sparse (low percent cover) and stunted vegetation, mosquito ditches, fragmented high marsh vegetation, elevation deficits, and minimal faunal usage of pools.

Target dredged material placement elevation—The initial elevation up to which dredged material is placed on the habitat enhancement site (e.g., marsh plain). After placement, the material undergoes predicted dewatering and consolidation of the dredged material, sinking from the target dredged material placement elevation to achieve the Target Ecological Elevation.

Target Ecological Elevation—The elevation necessary to meet the specific habitat (e.g., marsh) enhancement ecological objectives for a project. The Target Ecological Elevation is lower than the target dredged material placement elevation and is based on biological benchmarks, desired hydrology, and other conditions at the project site.

Thin-layer placement (TLP)—A wetland enhancement method in which dredged material (i.e., sediment) is intentionally placed on a wetland to increase its elevation, enhancing its resiliency while maintaining the hydroperiod necessary for native wetland vegetation. The thickness of the material is limited to enable the vegetation to grow back through the placed dredged material. Wetland ecosystems are expected to recover more quickly after TLP than after thicker applications of dredged material.[2]

[2] The U.S. Army Corps of Engineers defines TLP as "Purposeful placement of thin layers of sediment (e.g., dredged material) in an environmentally acceptable manner to achieve a target elevation of thickness. Thin layer placement projects may include efforts to support infrastructure and/or create, maintain, enhance, or restore ecological function." (Berkowitz et al., 2019)

The funding provided by the NFWF grant (Grant #43095) and the EPA grant (Grant #96273100) allowed the Project Team to develop a scope of work that incorporated a wide variety of monitoring and assessment activities. As such, project assessment included (1) the development of a monitoring plan that included flexibility for adaptive management monitoring, (2) data evaluation, (3) an analysis of key project costs, and (4) an analysis of the project's potential impact on community resilience. Most of the monitoring activities were focused on the marsh enhancement portions of the project, as well as avian responses to the creation of the elevated nesting habitat (ENH) on Ring Island. Protocols for these activities can be found on the New Jersey Department of Environmental Protection's Division of Science and Research website[3]. Contact Metthea Yepsen at Metthea.Yepsen@dep.nj.gov to request access to monitoring reports with detailed results and data.

The results presented in this report represent the responses of the marsh ecosystem between three to seven years after the placement of dredged material. The marsh enhancement activities were inherently disruptive and, thus, caused initial impacts on the ecosystem. This was expected, and continued monitoring is needed to track the ecosystem's trajectory and to determine if the success criteria of the projects have been achieved.

Detailed project summaries can be found in *The Nature Conservancy and New Jersey Department of Environmental Protection (2020). Beneficial use of dredged material to Enhance Salt Marsh Habitat in New Jersey: Project Summary and Lessons Learned on Project Development and Implementation.*[1]

[3] https://www.nj.gov/dep/dsr/wetlands/

Summary of Projects

Between August 2014 and April 2017, **enhancement** pilot projects were constructed at two locations – Ring Island and Avalon – in the Cape May Wetlands Wildlife Management Area and at a third location in the Fortescue Wildlife Management Area in Fortescue (Cumberland County; Figure 1). The marsh enhancement techniques included **thin-layer placement** (**TLP**) of dredged material on the platform of a vegetated, **stressed marsh** (Ring Island, Avalon, and Fortescue) and the filling of degraded and expanding pool-panne complexes with TLP on the surrounding stressed marsh platform (Avalon). Additional project components included the creation of **Elevated Nesting Habitat (ENH)** for birds at Ring Island and dune restoration and beach restoration projects at Fortescue. Table 1 shows a summary of the projects.

Figure 1. *Pilot beneficial use of dredged material projects were implemented at three locations in Cape May and Cumberland Counties, New Jersey.*

Table 1. *Summary of the pilot projects at Ring Island, Avalon, and Fortescue.*

Project	Ring Island		Avalon		Fortescue		
	Marsh	**ENH***	**Phase 1**	**Phase 2**	**Marsh**	**Beach**	**Dune**
Constructed	Aug 2014	Aug 2014	Dec 2014 to Jan 2015	Nov 2015 to Feb 2016	March 2016	Mar to Apr 2016	Feb to Apr 2017
Habitat Size (acres)	0.89	1	6.9	45	6.6	1.3	2.25
Sediment Volume (CY)**	1,000	6,000	~6,000	~49,300	6,490	7,245	18,335
Sediment Type	96% fine sand	96% fine sand	65% silt/clay	72% silt/clay	30% silt/clay	>80% Sand	>90% Sand
Placement Technique	Spray	Direct pumping	Spray	Spray & Direct pumping	Direct pumping	Direct pumping	Direct pumping
*ENH: Elevated Nesting Habitat **CY: Cubic Yards							

Ring Island

The dredged material for the Ring Island project was 96% fine sand, which came from a shoaled area in the New Jersey Intracoastal Waterway (NJIWW) located next to the island. The sand was used to construct two TLP areas (~0.5 acres each) and an ENH (~1 acre).

Elevated Nesting Habitat

The purpose of the ENH at Ring Island was to create suitable habitat for the State-endangered black skimmer (*Rynchops niger*) and other bird species of concern (Figure 2). It was designed to be a 1-acre mound with a platform elevation of 6 feet above Mean Higher High Water (MHHW) and 1:12 (vertical to horizontal) side slopes.

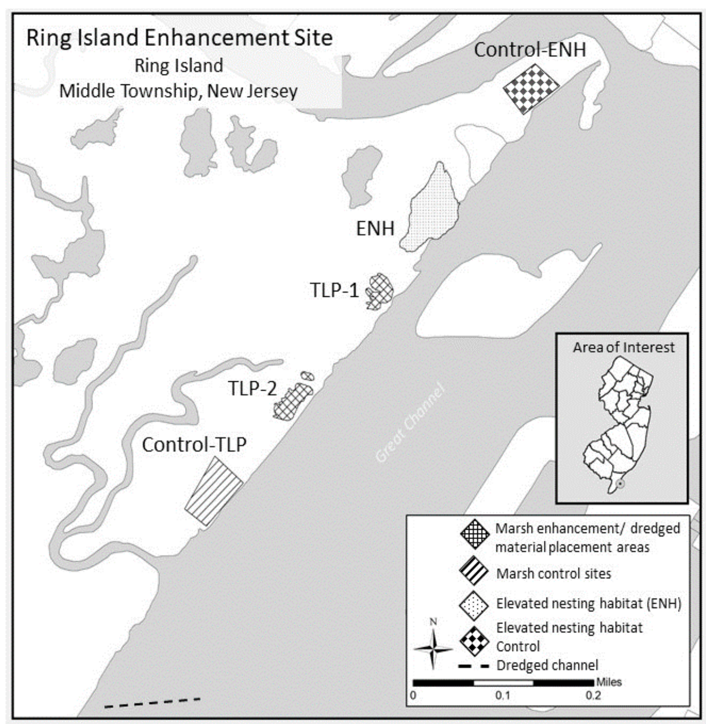


Figure 2. *Ring Island Project Site, showing the locations of the marsh enhancement/dredged material thin-layer placement areas (TLP-1 and -2), elevated nesting habitat and associated Control areas (Control-TLP and Control-ENH).*

In August 2014, the 1-acre ENH was created from approximately 6,000 cubic yards (CY) of the dredged sand. To keep the dredged material within the boundary of the ENH, the area was mostly enclosed during placement (except for a small outlet area for water to exit) using straw bales, a sand berm, and a silt fence. A skid loader was used to grade the placed dredged material and form the ENH mound. The top of the ENH was above the spring high tide line – an important nest survival threshold for coastal birds – but below the target elevation in the design and the permit. Post-construction adaptive management included planting vegetation on the slopes of the site, removing *Phragmites australis* from the site, removing plants from the top of the platform, and in early 2018, placing additional dredged material to restore elevations above the spring high tide.

Marsh Enhancement

The goal of this component was to increase the abundance and rigor of native marsh vegetation by increasing the elevation of "low-lying"[4] marsh by spraying thin (3- or 6-inch-thick) layers of sandy dredged material. These placement depths were expected to allow existing vegetation to survive and quickly recover, growing through the placed dredged material (Ray 2007). No perimeter containment was used because the surrounding areas of the marsh platform were at higher elevations than the placement areas and the sand did not disperse far during placement.

Over a period of two days in August 2014, approximately 1,000 CY of fine-grained sand were sprayed on two 0.5-acre sections of marsh platform using a hydraulic dredge and a spray nozzle system. The discharge end of the dredge pipe was placed on a pontoon at the edge of the marsh, with the spray landing approximately 150-200 feet from the pipe. Spraying the sand to achieve an even thickness of placement was difficult as it accumulated wherever it landed, with little natural spreading. Dredging was stopped intermittently, and water was sprayed through the nozzle system in an unsuccessful effort to disburse placed sand across the marsh platform.

Topographic surveys and depth measurements made in the weeks immediately after construction indicated that, although the average placement depth (6 inches) was within the 3- and 6-inch targets, placement was uneven, ranging between 0.5 to 9 inches. In March 2017, half of the bare areas within each of the TLP areas were experimentally planted with native salt marsh species.

Avalon

The Avalon Project site consisted of two phases of dredged material placement. The dredged material for these projects came from the NJIWW located near the project site; following Superstorm Sandy, this stretch of the NJIWW was one of the critical channel shoals that the USACE-OP needed to dredge. The sediment in the channel was predominantly silt and clay.

---

[4] The location of the possible TLP sites on Ring Island changed several times in the weeks preceding placement due to the presence of nesting birds and a change in the dredging schedule. As a result, the baseline topographic data could not be used to quantitatively assess the elevation of the selected TLP areas. However, based on observations during site visits, the selected TLP areas were lower than the surrounding and more densely and diversely vegetated, marsh. The Project Team decided it was worth moving forward with the project as an experiment and learning experience. Based on assessments conducted after construction, the baseline elevation of the TLP areas was closer to Mean High Water than Mean Higher High Water and 0.75 feet below the lower end of the range in elevation at which *Spartina patens* was found at the site.

Building upon the Ring Island project, the goal of the Avalon marsh enhancement project was to increase the area of vegetated marsh by filling expanding and degraded pools and pannes to the elevation of the surrounding marsh. The newly created topography was also expected to improve the drainage of the marsh, reducing excessive ponding of water, which is a primary stressor to plants. To accomplish this, the dredged material was either sprayed or directly pumped into the pools, with the overflow resulting in a thin layer of dredged material on the adjacent marsh platform. To keep the fine-grained dredged material within the placement area boundaries, perimeter containment (coconut-fiber logs) was installed before construction. Containment was added as needed during construction.

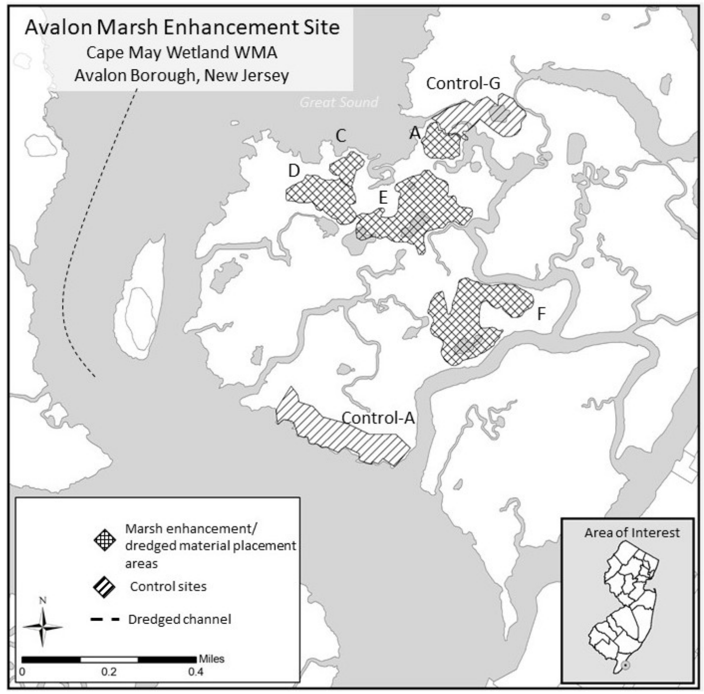


Figure 3. *Avalon Project Site, showing the locations of the marsh enhancement/Dredged Material Placement Areas (A through F) and marsh Control areas (Control-A and -G).*

Avalon Phase 1

In early January 2015, approximately 5,000 CY of predominantly fine-grained sediment were placed on 7 acres of the marsh (Areas A and C in Figure 3). The goal was to fill pools and add 3-6 inches of sediment to the surrounding marsh platform. In addition, sediment was sprayed directly on the marsh platform to learn how TLP with fine-grained material differed from sandy material.

Avalon Phase 2

Additional data collection and detailed engineering work were completed to design the Avalon Phase 2 project. Topographic surveys, high marsh **biological benchmarks,** and tidal data were used to establish **target dredged material placement elevations** and **Target Ecological Elevations**, rather than simply establishing target dredged material placement thicknesses. Between November 2015 and March 2016 about 50,000 CY of fine-grained dredged material from the nearby NJIWW navigation channel were placed on approximately 45 acres of the Avalon marsh (Areas A, C, D, E, and F in Figure 3).

Topographic surveys in the weeks following sediment placement showed the mean elevation of each placement area was within 4 inches of the target dredged material placement elevations. However, based on the range in these elevations, it is clear that sections of each placement area did not achieve the target dredged material placement elevations, while in other areas, the targets were exceeded by more than 1 foot. The depth of sediment placed on the marsh platform ranged from 1- 9 inches (placement was thicker in pools). Two years after placement, the areas lost some of their initial elevation and the mean elevation was within 1 inch of the Target Ecological Elevations at four of the five placement areas. However, some pools had reformed by 2017, indicating differential settling, consolidation, and compaction of the placed dredged material and underlying marsh.

After construction of the Avalon Phase 2 project, an adaptive management program was implemented. For example, based on observations that the dredged material was still dewatering and elevations were changing, combined with concerns regarding the potential to disrupt nesting birds, planting was delayed for a year. The original planting plan was also adaptively implemented in the field in response to the changed conditions. In addition, during the first summer after construction, vegetation die-off was observed in some areas outside and directly adjacent to the perimeter containment. Supplemental monitoring was conducted to investigate the cause(s) of the die-off and to suggest potential adaptive management actions to mitigate it. Initial results from this monitoring suggested that the containment was blocking tidal flow and preventing dewatering of the sediment. This, in combination with acid produced by the oxidation of the placed dredged material, may have created extreme water chemistry fluctuations (e.g., extremely high pH and salinity in surface waters) and less-than-optimal flooding patterns. This led to the removal of most of the containment at both Avalon and Fortescue rather than continuing with the original plan for it to biodegrade in-place.

Fortescue

The marsh enhancement component of the Fortescue pilot project was designed to improve growing conditions for native salt marsh plants and create positive drainage of the marsh platform by increasing the elevation of a low-lying ditched marsh. In addition, to protect the marsh and nearby marina from erosion, a dune restoration project was designed, and a nearby eroding beach was restored (Figure 4). To construct these projects, mostly sandy sediment from the nearby Fortescue Creek navigation channel was dredged.

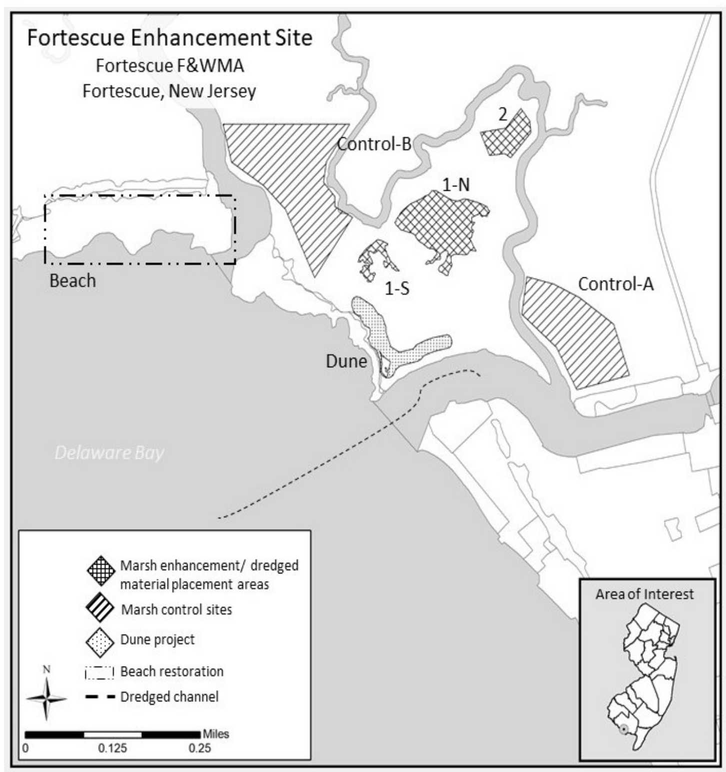


Figure 4. *Fortescue Project Site, showing the locations of the marsh enhancement (1-S, 1-N, 2), dune restoration (Dune) and beach nourishment (Beach) Dredged Material Placement Areas and the marsh Control sites (Control-A and Control-B).*

Marsh Enhancement

The Fortescue project was designed to increase the elevation of the marsh by approximately 9 inches. The target dredged material placement elevation of 3.3 feet and the Target Ecological Elevation of 2.8-3.0 feet NAVD88 were based on biological benchmarks and local site hydrology with the objective to produce high marsh habitat. The Project Team applied the lessons learned from the construction of the Avalon Phase 2 project to the design and construction of the Fortescue project. The marsh enhancement design included a branching system of pipes with valves that could be opened and closed and included some flexible pipes that could be moved more easily by hand. This design would, in theory, avoid the costly downtime that was experienced during the Avalon Phase 2 project when the discharge of dredged material had to be stopped to prevent overflow of the containment and when the dredge pipe had to be repositioned. A double containment row of plastic mesh tubes filled with wood chips (Filtrexx SiltSoxx™) was installed around each Dredged Material Placement Area.

In March 2016, 6,490 CY of a heterogeneous mix of sandy and fine-grained dredged material from the Fortescue Creek navigation channel were placed on two areas, totaling 6.4 acres of the marsh, which was less than 30% of the originally planned marsh enhancement area. The efficacy of the highly engineered dredge pipe network could not be evaluated because it was not used to its full advantage, and large machinery was used to move the position of the pipe outlet rather than employing the smaller flexible pipes. Several challenges were encountered during the marsh enhancement. For example, the dredged material was sandier than expected, the winter weather was especially harsh and there were contracting issues. As a result, the Project Team was unable to complete construction in the first year and decided not to finish the project in the second year. A survey of dredged material depth in permanent monitoring plots found that the average depth of placement was 6 inches and ranged from minimal to 18 inches.[5]

After construction, portions of the bare sections of the marsh enhancement areas were planted with native salt marsh species. As part of the project adaptive management plan, the plastic netting around the Filtrexx SiltSoxx™ was removed and the woodchip filling was dispersed across the marsh.

Dune Restoration

The Fortescue dune restoration was designed to protect the adjacent marsh and the marinas on Fortescue Creek from erosion caused by tidal flow and storm waves. Originally, this project component was designed to be constructed mainly in the footprint of an existing dune adjacent to the marsh platform, but after a winter storm eroded portions of the shoreline the dune footprint was redesigned to be further in the marsh.

In early 2017, approximately 18,000 CY of sandy dredged material were used to construct the dune. Containment made from Filtrexx SiltSoxx™ and sand berms was used to retain the dredged sand. Outlets through the containment on the marsh side of the dune were constructed to allow fine-grained sediment to flow into the marsh creating an area of TLP.

[5] While RTK transect surveys of the elevation at Fortescue were conducted each year, the Control Areas showed improbable gains in elevation. For this reason, elevation data is not summarized here.

“As built” surveys show that the final constructed dune was 900 feet long, 40 feet wide at the top, and 80 feet wide at the bottom, with an elevation of 10.0 feet NAVD88 (5-6 feet above the marsh surface). After construction, the dune was planted with native shrubs and grasses.

Beach Nourishment

The Fortescue project’s beach nourishment component was designed to restore a natural beach near Fortescue Creek. The beach was eroded by winds and waves. Approximately 7,000 CY of sediment (at least 80% sand) dredged from the Fortescue Creek navigation channel were spread over 1.3 acres (700 linear feet) of beach at a 15:1 (horizontal to vertical) slope. It was expected that a plateau would form naturally as the placed dredged material was "worked" by the tides and wind.

Monitoring Plan: Objectives

As the first of their kind in New Jersey, the salt marsh enhancement projects tested the idea that the application of dredged material (i.e., sediment) on existing, but stressed and vulnerable salt marshes can result in ecological enhancement and help them persist into the future in the face of sea-level rise, erosion, and subsidence. With three different project sites, the Project Team was able to test a variety of dredged sediment types, placement methods, and sediment depths on a range of baseline conditions. In the long run, the projects will be considered a success if there is (1) an increase in and maintenance of an optimal tidal elevation and hydroperiod for native salt marsh biota, (2) an increase in the abundance and vigor of native salt marsh plants, and (3) a return for all other parameters to pre-construction conditions unless they were expected to change from the conversation of habitat.

The Ring Island Elevated Nesting Habitat (ENH) was designed to create nesting habitat primarily for the black skimmer (*Rynchops niger*), a State-endangered species, and other colonial and marsh nesting species. The project would be considered a success if (1) the increased elevation kept the nests from being flooded, (2) the vegetation in the sandy ENH habitat remained at densities low enough to maintain suitable nesting habitat and was not colonized by invasive plants (for example, *Phragmites australis*), and (3) black skimmers and other colonial and marsh nesting species were successful in fledging chicks from the habitat. Due to the unexpected use of this habitat by other species of interest, the definition of success for the ENH project has expanded beyond its original avian targets to include other species of sandy habitat nesters including horseshoe crab (*Limulus polyphemus*) and the diamondback terrapin (*Malaclemys terrapin*).

The beach restoration and dune enhancement projects at Fortescue were primarily evaluated based on the consistency of their “as built” condition with their design. In addition, goals for the enhanced dune included (1) the successful survival and establishment of native plantings, (2) the removal and prevention of future establishment of *P. australis*, and (3) low rates of erosion.

Several different types of monitoring were conducted during the implementation of the projects. Baseline data collection before project construction (i.e., placement of dredged material) was used to characterize existing conditions and inform final site selection and project design; “as built” surveys were conducted to determine if the projects were built as designed. A formal Monitoring Plan was created in 2014 and implemented to track progress toward meeting the ecological goals of the projects. HAZUS modeling was used to estimate the potential socioeconomic benefits of the Avalon and Fortescue projects (Ferencz et al., 2017). Adaptive management monitoring addressed several issues identified during post-construction monthly site inspections.

Monitoring Plan: Data Evaluation

The Monitoring Plan was devised to track the initial responses (one to three years post-construction) of a variety of ecological salt marsh parameters to dredged material placement, as well as to track the progress/trajectory toward meeting the project goals; thus, it is purposefully comprehensive. The Monitoring Plan and subsequent modeling and adaptive monitoring were designed to answer research questions to help inform future projects of this kind and to help determine the feasibility of scaling up this work within the state. As such, all project work was carefully documented, and project sites continue to be monitored.

The questions that the team hoped to answer through monitoring included:

- Can dredged material be placed on a marsh consistent (vertically and horizontally) with the project's design?
- Were Target Ecological Elevations appropriate for developing native high marsh plant communities?
- How much does the placed dredged material consolidate over time?
- Does the marsh ecosystem (i.e., structure and functions) recover or show uplift within two to three years of dredged material placement? (And the related "How long does it take for the marsh to recover and achieve uplift?")
- Did differences in structural factors (e.g., grain size, elevation, placement thickness) correlate with recovery and uplift?
- Were there any unexpected outcomes of the placement of dredged material?
- What parameters should be included in a marsh enhancement monitoring program (before, during, and after placement)? And how long should the post-placement program be conducted?

The monitoring activities used widely accepted and scientifically defensible methods. Where possible, these methods were adapted from the Salt Marsh Integrity Index Protocols used by the U.S. Fish and Wildlife Service (USFWS) Region 5 (Neckles et al. 2013). The USFWS New Jersey National Wildlife Refuges planned to use these protocols for their beneficial use of dredged material projects, and the Project Team intended the data to be comparable. When possible, monitoring activities took place before and after placement of dredged material and in a control site, following a Before-After-Control-Impact (BACI) design. "Impact" portions of the project are typically referred to as dredged material "placement areas." The elements of the Monitoring Plan are listed in Table 2. Additional research on sediment and water chemistry characteristics at Avalon and regular site inspections at project sites were added after the dredged material had been placed as part of an adaptive monitoring program; these activities proved to be very helpful in adaptively managing the projects.

***Table 2**. Gantt chart of monitoring categories. Red coloration signifies the year that dredged material was placement within the site. Green represents that monitoring of the specific metric occurred within the year. Monitoring was performed generally during peak biomass season (summer to fall).*

	Metric	2013	2014	2015	2016	2017	2018	2019	2020	2021	2022
Ring Island	Placement										
	Plant Community										
	Benthic Invertebrate Species										
	Avian Use										
	Habitat Change Analyses										
	Site Visits										
	Surface Elevation Tables										
	Topographic Surveys										
	Sediment Characteristics										
	Water Level										
	Nekton										
	Water Chemistry										
Avalon	**Metric**	2013	2014	2015	2016	2017	2018	2019	2020	2021	2022
	Placement										
	Plant Community										
	Benthic Invertebrate Species										
	Avian Use										
	Habitat Change Analyses										
	Site Visits										
	Surface Elevation Tables										
	Topographic Surveys										
	Sediment Characteristics										
	Water Level										
	Nekton										
	Water Chemistry										
Fortescue	**Metric**	2013	2014	2015	2016	2017	2018	2019	2020	2021	2022
	Placement										
	Plant Community										
	Benthic Invertebrate Species										
	Avian Use										
	Habitat Change Analyses										
	Site Visits										
	Surface Elevation Tables										
	Topographic Surveys										
	Sediment Characteristics										
	Water Level										
	Nekton										
	Water Chemistry										

1. Horizontal Extent of Dredged Material Placement

Jackie Jahn[1,2], Lisa Ferguson[3], Metthea Yepsen[2,4]

[1]GreenVest, [2]The Nature Conservancy, [3]The Wetlands Institute, [4]New Jersey Department of Environmental Protection

At each site, areas were selected for placing sediment based on several criteria and a variety of mechanisms were used to attempt to get sediment spread into and across those areas. Details are discussed in the 2021 Beneficial Use of Dredged Material to Enhance Salt Marsh Habitat in New Jersey: Project Summary and Lessons Learned report.[6]

Monitoring Design

Extent of Placement

The extent of sediment placed in marsh enhancement areas was mapped by GreenVest. The perimeter of placed sediment areas was walked shortly after placement with a Global Positioning System (GPS) to create placement polygons. Polygons were imported into ArcGIS ArcMap (Version 10.4) and used to calculate acres of placement area.

Results

Ring Island TLP

It was difficult to spray dredged material into the "inland" corners of the rectangularly delineated placement areas, and spraying had to stop once the maximum elevations had been reached in some areas; as a result, dredged material was placed on 0.89 acres of the targeted 1- to 1.4-acre TLP area (Figure 5).

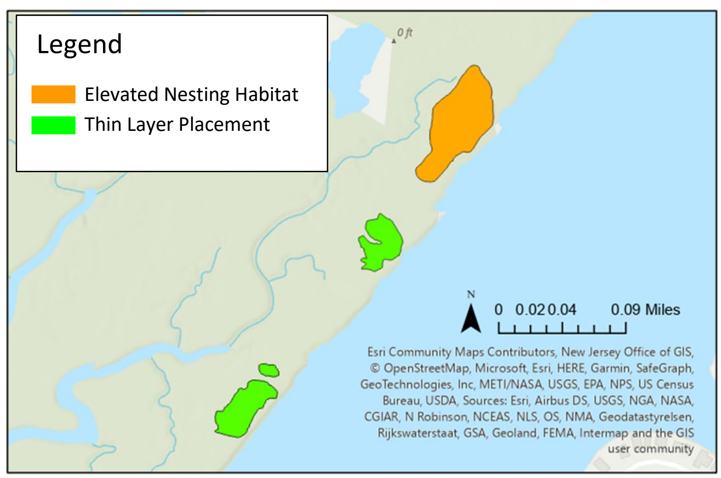


***Figure 5.** Placement extent the month after construction at Ring Island.*

Ring Island ENH

The planned footprint of the ENH was 1.14 acres in size, and the footprint of the constructed projects was 1.49 acres one month after placement.

The extent of the ENH suitable for nesting by the target species changed over time (see Figure 6). This "nesting platform" was defined as the contiguous area above the Spring Tide level. At the time of placement (September 2014), the platform was 0.9 acres. As the platform was high with somewhat steep

[6] https://www.nj.gov/dep/dsr/wetlands/beneficial-use-dredged-material-project-summary-lessons-learned.pdf

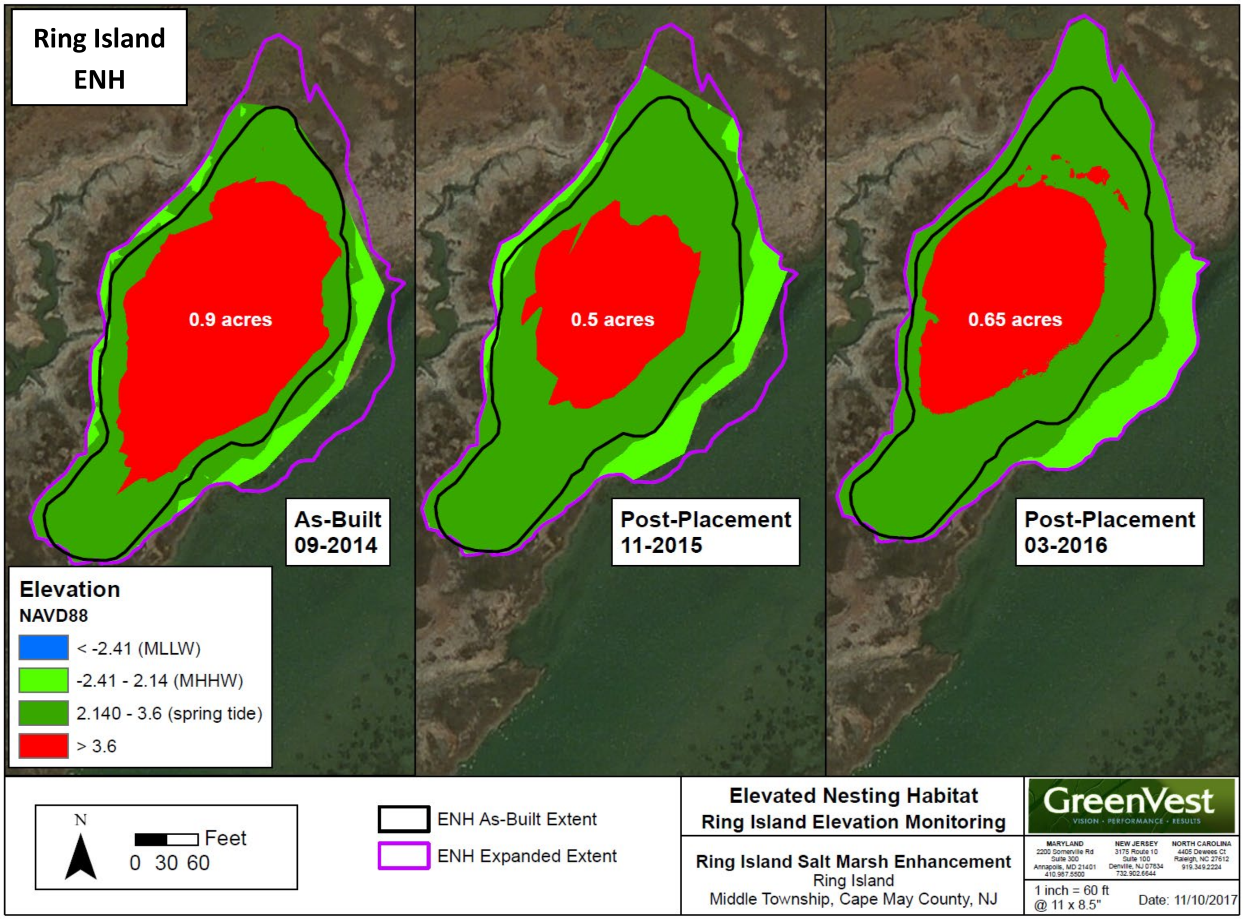


Figure 6. *Change in the shape and elevation of the Elevated Nesting Habitat at Ring Island from September 2014 to March 2016. Surveys from 2014 and 2015 were conducted using RTK-GPS and the survey from March 2016 was conducted using ground-based LiDAR. The acreage for the nesting platform was measured to be 0.9 acres in 2014, 0.5 acres in 2015, and 0.65 acres in 2016. The acreages correspond only to the nesting platforms and not the overall placement area.*

slopes, it was expected that wind and water forces would reduce platform elevation. By November 2015, the platform reduced to 0.5 acres and by March 2016, it expanded slightly to 0.65 acres. This increase may be due to the difference in survey methods between 2015 and 2016 and/or a result of sand moving from the platform onto surrounding areas.

Although the overall nesting platform reduced in size by 2016, the remaining platform was found to be heavily utilized by nesting birds and vegetation began to establish. However, the ENH had lost a significant amount of elevation (on average 0.36 ft NAVD 88; see Section 3: Elevation). Therefore, the project team decided to create a larger nesting platform through the addition of more dredged material in early 2018.

Avalon

43.9 acres of the planned 51 acres of marsh received dredged material (Figure 7). Placement areas that received sediment were larger than planned due to sediments overtopping containments. Additionally, sediment placement took longer than expected, and the project ran out of time before all placement areas received sediment.

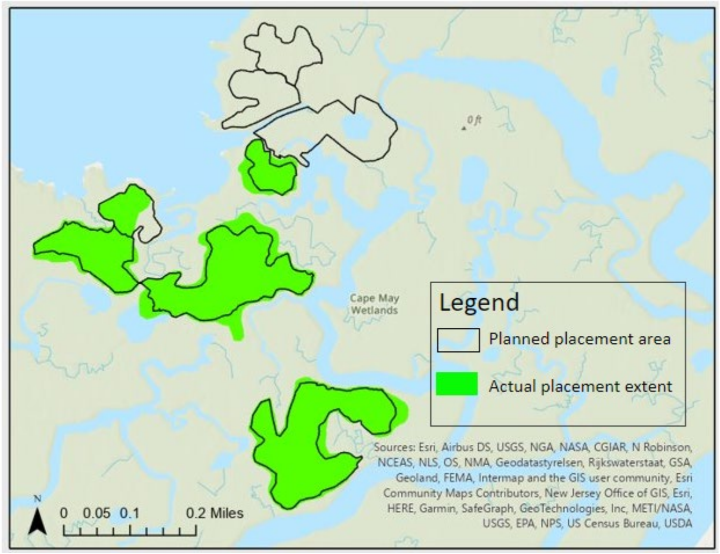


Figure 7. *Planned vs actual marsh project at Avalon.*

Fortescue

6.5 of the targeted 23 acres received sediment in late winter of 2016 (Figure 8). NJDOT intended to return and place more sediment at the site but was unable to do so due to bad weather in the spring of 2016 and courser than expected sediment. Figure 8 depicts areas planned for placement and areas that received dredged sediment.

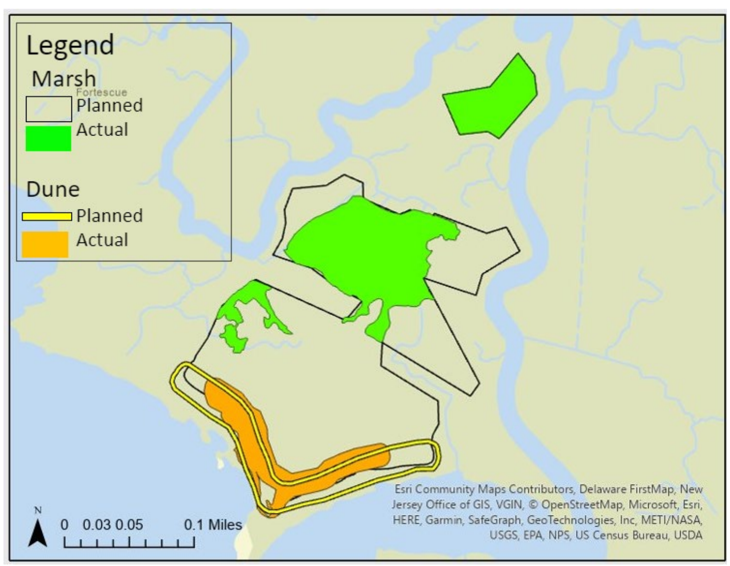


Figure 8. *Planned vs actual marsh and dune projects at Fortescue.*

Conclusions

- All projects ended up being smaller than planned due to time constraints and difficulty spreading the heavy sediments.
- At Ring Island, it proved too difficult to spray sand in an even layer or a rectangular shape. As a result, the team had to stop pumping before it could reach the corners and interior portions of the placement area, which were more difficult to reach.
- At Avalon, the individual placement areas were larger than planned because sediment overflowed the containments, and the team ran out of time to place sediment in all planned areas.
- At Fortescue, there were a variety of factors that led to a smaller project, including bad weather and difficulty spreading sandy dredged materials.
- Based on these analyses, we can answer the following monitoring question: Can dredged material be placed on a marsh consistent (horizontally) with the project's design?
    - We found the answer to be no, it is unlikely that the sediment can be placed on the marsh exactly according to design. Complications can arise during placement, such as difficultly moving the sediment and machinery needed, which can prevent complete coverage of a planned area.

2. Depth of Dredged Material Placement

Laura Moritzen[1], Jessie Buckner[1], Metthea Yepsen[1,2], Adrianna Zito-Livingston[1]
[1]The Nature Conservancy, [2]New Jersey Department of Environmental Protection

In addition to changing the elevation of the marsh, the depth of the dredged material placed on the marsh was expected to affect vegetation and animal species survival and the trajectory of post-placement recovery (see Vegetation Recovery Section). In cases where plants or animals were completely smothered, it was expected that recovery (and potentially ecological) uplift would take longer than areas of the marsh covered with a comparatively thin layer of dredged material. Previous research demonstrated that salt marsh plants can recover after the placement of sediment for elevation enhancement. For example, Ray (2007) found that plants recovered best after placement of 0.17-0.50 feet of material, and Reimold et al. (1978) found excellent vegetation recovery after placement of up to 0.75 feet of sandy material.

Monitoring Design

There are a number of different ways to measure the depth of dredged material placed on the marsh. The results reported here are based on two methods: 1) measurements taken by TNC in the permanent vegetation plots for two-years post-placement by piercing the dredged material with a ruler and measuring the depth down to the underlying original marsh surface/root mat (five measurements per plot; see Section 5: Vegetation from Plot Based Monitoring), and 2) digital elevation models (DEMs) based on topographic surveys (Table 3). Measurements of sediment depth in plots were likely more accurate than interpolated RTK-GPS points due to the error associated with RTKs as well as with interpolation of topographic data. However, the interpolation maps give a much better picture of the spatial variability of sediment depth and are thus still worth evaluating.

Table 3. *Design and metrics used for monitoring the depth of dredged material placement at Ring Island, Avalon, and Fortescue project sites.*

Metrics	Method	Design
Depth of dredged material placement (cm and ft)	Ruler measurements	Five measurements in each permanent vegetation plot during peak of growing season
Change in elevation (ft)	Digital Elevation Models (DEMs)	Interpolated RTK-GPS transect surveys or ground based LiDAR surveys.

Results

Ring Island TLP

The dredged material used at Ring Island was 96% sand and the average depth of placement across the site as of two-years post-placement was 15.0 cm (0.5 ft, SD = 5 cm or 0.2 ft). Although it was observed that individual plots experienced small increases or decreases in their average placement depths from two to three years post-placement, there was no significant net increase or decrease for the project (ANOVA, $p > 0.05$). Despite large ranges in elevation, a comparison of DEMs from the baseline and as-built surveys indicate that 64% of the site gained 7.62-15.34 cm (0.25-0.5 ft) of sediment, which matched the target depths set for the project. The spatial distribution of elevation changes between the baseline and the "as built" can be seen in Figure 9.

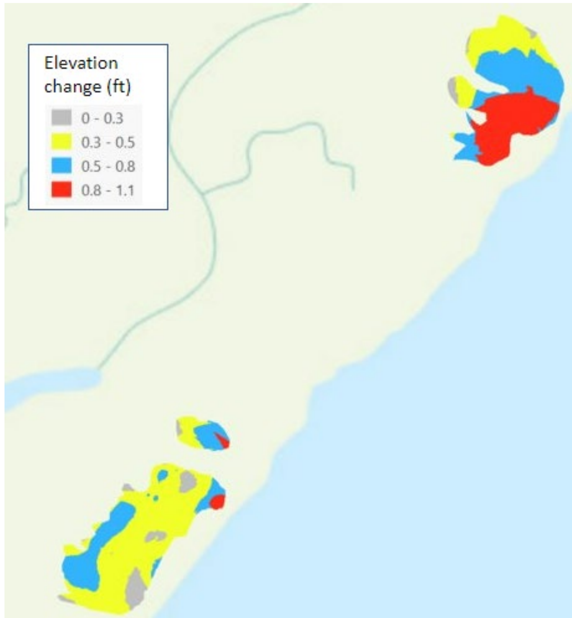


Figure 9. *Change in elevation (ft) between baseline and "as built" at Ring Island TLP areas.*

Avalon Phase 2

During the "as built" survey, the average depth of placement was 30.8 cm (1.0 ft, SD = 27.1 cm or 0.9 ft) for plots that started as marsh platform, and >62.6 cm (2.1 ft, SD = 37.3 cm or 1.2 ft) for plots that started as a pool (Figure 10). Sediment tended to stack up around pipe outlets and depth decreased moving away from pipe outlets. Two-years post-placement, these averages both decreased to 19.8 cm (0.7 ft, SD = 18.9 cm or 0.6 ft) and 42.9 cm (1.4 ft, SD = 37.4 cm or 1.2 ft), respectively. However, these decreases were not statistically significant (ANOVA, $p > 0.05$ platform; Kruskal-Wallis, $p > 0.05$ pool).

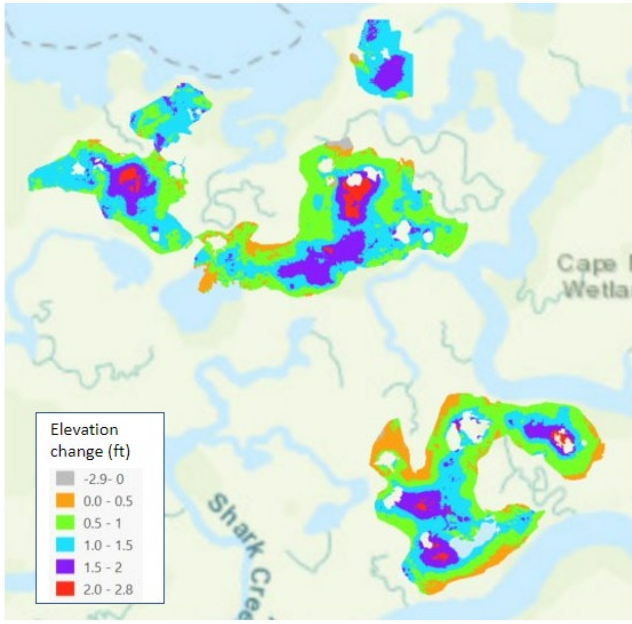


Figure 10. *Change in elevation (ft) between baseline and "as built" at Avalon in placement areas.*

Fortescue

The dredged material placed at Fortescue was a silt and sand mixture, and the average depth of placement measured in the vegetation plots one-year post-placement was 17.4 cm (0.6 ft, SD = 15.4 cm or 0.5 ft). This average decreased slightly, but not significantly, to 15.4 cm (0.5 ft, SD = 6.6 cm or 0.21 ft) two-years post-placement (ANOVA, $p > 0.05$; Figure 11).

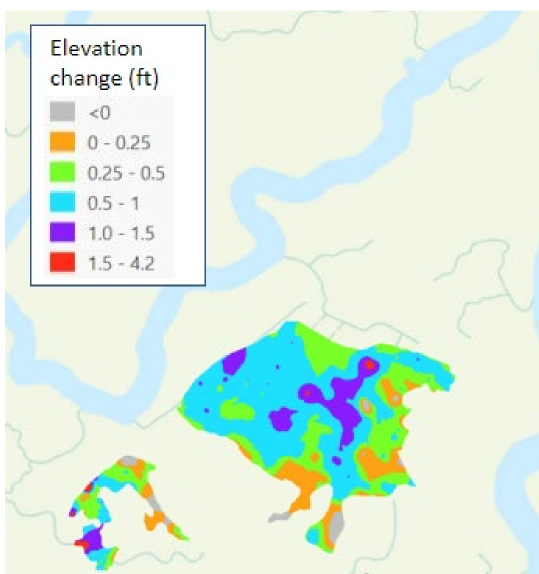


Figure 11. *Change in elevation (ft) in Fortescue TLP areas between baseline (2015) and "as built" (2016).*

Conclusions

- On average after two years post placement, dredged material depth at Ring Island and Fortescue was 0.5 ft, and at Avalon, it was 1 ft on the vegetated marsh platform and 2 ft deep in pools. However, there was a large range of placement depths at all sites.
- Sediment tended to stack up where it first made contact with the ground rather than spread into an even surface. This was even true at Avalon where the sediment was primarily fine-grained. The effect of dredge material depth is explored in Section 5: Vegetation from Plot Based Monitoring below. It is possible that this variability is not ecologically detrimental and instead may lead to a more diverse habitat with niches for a variety of native salt marsh species.
- Based on these analyses, we can answer the following monitoring question: Can dredged material be placed on a marsh to achieve a consistent elevation?
  - We found the answer to be likely no. Due to issues with equipment and sorting of the sediments during application, the sediment was not placed at an even depth across the marsh platform. Pools and ponds resulted in additional differences in depth due to the high rates of compaction and vast depths of the ponds themselves.
  - The dredged material was generally placed at the Target Dredged Material Placement Elevations in around 50% of each site. The remainder of the site was generally higher than planned at Ring Island, lower than planned at Fortescue, and both above and below at Avalon.

3. Elevation

Metthea Yepsen[1], GreenVest
[1]The New Jersey Department of Environmental Protection

Contributors: Nick Procopio[1], Lori Lester[1]

The primary objective of the placement of dredged material on the Ring Island, Avalon, and Fortescue salt marshes was to increase the elevation of the marsh surface to reduce flooding and change the marsh topography. Changes in the elevation and topography in a salt marsh influence hydrology, which, in part, determines the floral and faunal communities that are integral to the ecological structure and functions of the marsh. To attribute changes to the marsh ecosystem to the addition of dredged material, it is first imperative to accurately document elevation changes in project sites and control sites. In addition, as pilot projects, there was an interest in seeing how closely the projects were built to their design. Observed changes in elevation should be consistent with the depth of the placed dredge material under the assumptions that there is minimal subsidence of the marsh and that data collected using different methods are comparable.

Monitoring Design

Table 4. *Design and metrics used to monitor elevation, topography, and accretion at all three project sites.*

Metrics	Method	Design
Elevation (Ft NAVD88)	LiDAR or RTK-GPS	Minimum of baseline, “as built”, and one-year post construction and 2019 in placement and control sites.
Elevation (mm)	Shallow and Deep Surface Elevation Table	Installed post-placement. 3 per control site and 3 per placement site. Read annually in late summer.
Net Accretion (mm)	Marker Horizon Plots – Feldspar Plots and Sediment Grids	Installed post-placement. 4 per SET. Read annually in late summer.
Shallow Subsidence (mm)	Surface Elevation Tables combined with Marker Horizon Plots	Calculated annually.

Elevation surveys were conducted using Real-Time Kinematic-Global Positioning Systems (RTK-GPS) and ground-based Light Detection and Ranging (LiDAR) over five to six years, depending on the placement site (Table 4). Surveys before placement documented baseline conditions and informed placement targets. Surveys immediately post-placement (“as built” survey) were used to determine the thickness (depth) of dredged material placement and how similar the new elevations were to those targeted by the restoration design. Surveys conducted after the “as built” documented how the elevations changed as sediments consolidated, shifted, and compressed the marsh below it. Measurements in Years 1 and 2 post-placement were used to develop planting plans. Measurements in control sites were used to document natural variation and the sampling method accuracy. Measurements of pre- and post-placement elevations were compared with tide range, tidal amplitude, hydroperiod, and key biotic parameters to evaluate the efficacy of using dredged material for salt marsh enhancement. Although not well documented throughout the project, the vertical accuracy of the RTK-GPS units was set at 0.16 feet. As such, changes in elevation under 0.16 feet were assumed to be too small to measure.

Shallow and deep surface elevation tables (SETs) with marker horizons were installed post-construction by The Nature Conservancy (TNC) in control and placement areas. SETs are fine-scale instruments designed to capture changes in elevation (at the scale of millimeters) over time and help to explain the processes responsible for observed elevation changes (https://www.pwrc.usgs.gov/set/), including the accretion of sediment. To account for a heterogeneous, dynamic marsh surface and the noise within the data, SET data must have a minimum of three to five years of data that need to be collected before elevation changes are analyzed. At the time this report is being written, each site has at least six years of data accumulated for the SETs and associate marker horizon plots. These data are currently being analyzed and will be posted in the NJ Tidal Wetland Monitoring Network (NJTWMN) database[7].

Topographic surveys, data interpolation, and data interpretation were worked on by many groups including USACE, NJDOT, GreenVest, Princeton Hydro, TNC, Delaware Valley Data Collection, GBA, and NJDEP.

Results

Ring Island TLP

Based on studies by Ray (2007) and Reimold et al. (1978) that demonstrated plants could survive sediment placement, TLP Area 1 and TLP Area 2 at Ring Island were designed to receive 0.25 feet and 0.5 feet of dredged material, respectively. No statistical difference was found in the depth of placement between the two areas (0.4 ± 0.1 ft and 0.5 ± 0.1 ft, respectively), so they are being analyzed together.

Ring Island Elevation

Surveys at Ring Island were conducted using RTK-GPS and ground-based LiDAR over six years. Low marsh at Ring Island was defined as -0.40 to 1.69 feet NAVD88, and high marsh between 1.69 to 2.08 feet NAVD88 (see Section 4: Water Level Monitoring). Based on the RTK-GPS surveys, the elevation in the TLP areas immediately after placement increased by an average of 0.55 feet (Figure 12). The general elevation gain was maintained over the next five years or, in some locations, showed a slight increase (Figure 13B, Table 5). The measured median elevation in Ring Island control sites increased by 0.13 feet from baseline conditions in 2014 to 2019, five years post-construction. The measured median elevation in placement sites increased by 0.55 feet during the same period. By 2019, the median elevation in the placement sites was 0.25 feet higher than the control site (Kruskal-Wallis chi-squared, $p < 0.001$). Elevation gain measured in the control site was not greater than the vertical accuracy of the survey equipment (0.16 feet) and the sample size increased greatly from 2014 to 2019, thus, it was not possible to conclude that the median elevation increased at the control site (Table 5).

[7] The NJTWMN website and database are currently being finalized and have not been approved for public posting yet.

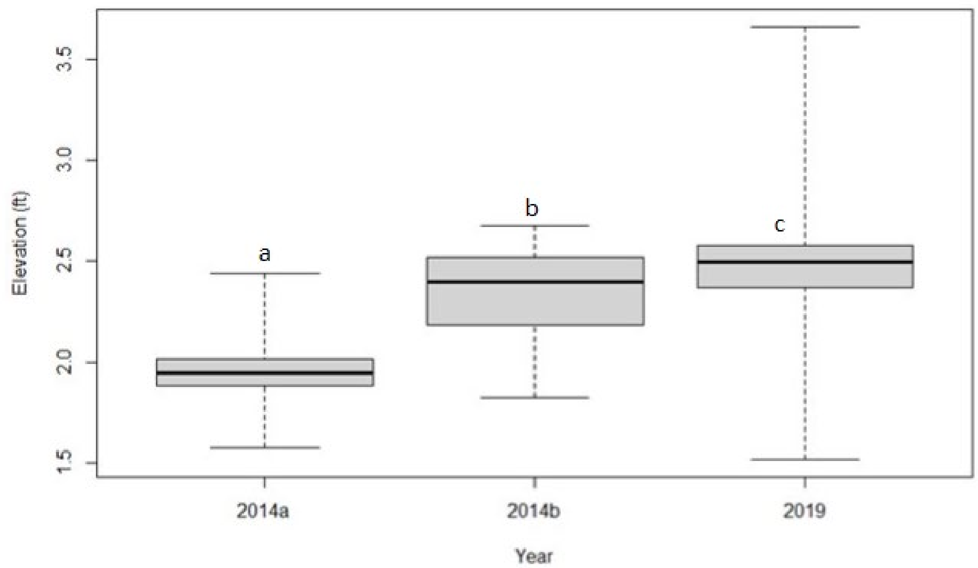


__Figure 12.__ Elevation (Ft NAVD88) quantiles at Ring Island placement areas based on RTK-GPS transect surveys at the site. Box and whisker plots include the minimum, quartiles, median, and maximum elevation. Elevations were statically different from year to year (Kruskal-Wallis chi-squared, p < 0.05). 2014a: n=46; 2014b: n=109; 2019: n=126).

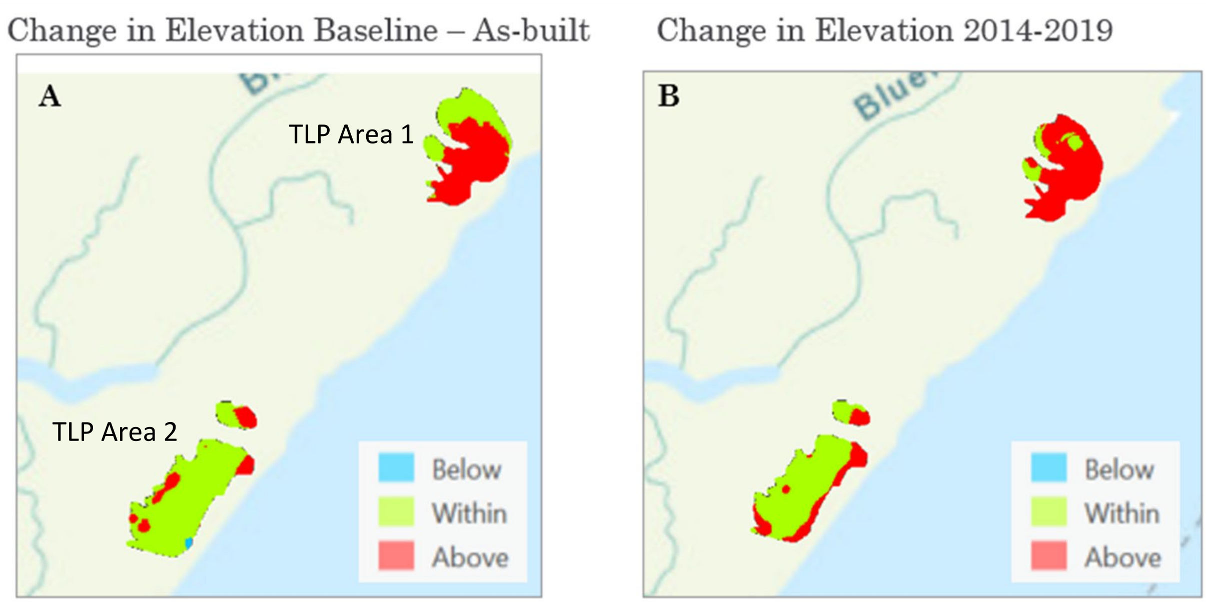


__Figure 13.__ Areas of Ring Island that were below, within, or above the 0.25-0.75 ft target elevation gains during the A) "as built" (2015) and B) five years post-placement surveys.

Table 6. *Elevation (Feet, NAVD88) quartiles at Ring Island in Control and Placement sites during baseline (2014) and five years post-placement in 2019 based on RTK-GPS surveys.*

		Elevation Quartiles (Feet, NAVD88)				
	n	**0%**	**25%**	**50%**	**75%**	**100%**
Control Baseline	7	1.90	2.05	2.12	2.17	2.19
Control 2019	150	0.06	2.13	2.25	2.34	3.31
Placement Baseline	50	1.65	1.90	1.95	2.02	2.44
Placement 2019	119	1.52	2.38	2.50	2.58	3.66

Ring Island Target Ecological Elevations

64% of the TLP area gained elevation between 0.25 and 0.75[8] feet of sediment, which was considered “within” target elevations (Figure 13A). The median elevation of the placement areas post-placement in 2019 was 2.5 feet NAVD88 (Table 5), which is above Mean Higher High Water elevation and above the upper-end of the elevations suitable for high marsh (see Section 4: Water Level Monitoring).

Ring Island Elevated Nesting Habitat (ENH)

The ENH was designed to add 5 feet of elevation above the existing marsh. This would place the top of the ENH 6 feet above Mean Higher High Water at 2.14 feet NAVD88. The “as built” survey showed that a large portion of the interior of the ENH placement area was above the spring high tide level (3.6 feet NAVD88), which is an important nest survival threshold for coastal birds. The results in Table 6 indicate that the high interior ENH platform (represented by the maximum elevations) lost 1.8 feet in elevation from the “as built” survey in the Fall of 2014 to the second post-placement survey in March of 2016, while the surrounding areas (assumed to be represented by the minimum elevations) gained 0.5 feet in

Table 5. *Elevation of the Elevated Nesting Habitat (ENH) at the Ring Island project site based on RTK-GPS and LiDAR surveys. No pre-placement data were collected.*

Elevated Nesting Habitat Elevation (Feet, NAVD88)				
	“As built” RTK survey Sept 2014	**Post-Placement RTK Nov 2015**	**Post-Placement LiDAR March 2016**	**Post-Placement LiDAR June 2017**
Minimum	0.27	0.22	1.38	1.62
Maximum	7.61	5.84	5.83	5.96
Range	7.34	5.62	4.45	4.34
Mean	**3.88**	**3.00**	**3.52**	**3.17**
Standard Deviation	1.49	0.90	0.71	0.77
n	279	176	NA	NA
Change in maximum elevation per year		-1.77	-0.01	+0.13
Change in mean elevation per year		-0.88	+0.52	-0.35

[8] An elevation buffer was added to the 0.25 to 0.50-foot target range to account for survey and interpolation error.

elevation. The loss of elevation from the interior platform was balanced largely by an increase in elevation of the surrounding platform area. The loss between the November 2015 and March 2016 surveys is indicative of transport and deposition during winter storms. By 2017, the losses in elevation of the interior ENH platform continued. In response, additional dredged material was placed on the interior portion of the ENH in February 2018.

Avalon Phase 2

The dredged material placement goals of the Avalon Phase 2 marsh enhancement project were to fill expanding pools to an elevation slightly above the surrounding vegetated marsh plain, with the areas around the pools receiving a thin layer placement of dredged material. Target dredged material placement elevations were set for each placement area based on the existing topography of the area and expected dewatering and consolidation of the dredged material (Table 7; see Section 4: Water Level Monitoring for further details on high and low marsh range elevations). This is the elevation up to which dredged material was planned to be placed. After placement, the dredged material underwent predicted dewatering and consolidation, sinking from the target dredged material placement elevation to achieve the Target Ecological Elevation. While Target Ecological Elevations are the elevation necessary to meet the specific habitat enhancement ecological objectives for a project, Target Ecological Elevations are lower than the target dredged material placement elevation and are based on biological benchmarks, desired hydrology, and other conditions at the project site.

Table 7. *Target elevations for placement areas at Avalon Phase 2.*

	Placement Area Elevation Targets (Feet, NAVD88)				
High Marsh Range	2.03 to 2.39				
Low Marsh Range	-0.03 to 2.03				
Placement Areas	**A**	**C**	**D**	**E**	**F**
Placement Target	3.00	2.61	3.00	2.39	3.00
Ecological Target	2.50	2.11	2.50	1.89	2.50

Avalon Elevation

Surveys were conducted using RTK-GPS and ground-based LiDAR over six years[9]. Based on interpolated RTK-GPS surveys conducted during the baseline and interpolated ground-based LiDAR surveys conducted in 2016 after sediment was placed, the initial “as built” increase at Avalon in mean elevation was between 0.90-1.23 feet across all five placement areas. Minimum, maximum, and mean elevations were increased in all dredged material placement areas and the variability in elevation decreased in all placement areas when compared with the baseline.

Median elevations in control and placement sites were statistically different in the baseline (Kruskal-Wallis chi-squared = 2726.7, df = 143, $p < 0.001$), but at 0.17 feet difference, just above the error associated with the RTK-GPS measurements (0.16 ft; Figure 14A and Table 8). The elevation difference between placement areas and control sites significantly increased to 0.27 feet five years after placement

[9] An elevation buffer was added to the 0.25 to 0.50-foot target range to account for survey and interpolation error. dditionally, the LiDAR surveys were not corrected for vegetation and may over-estimate elevation as a result. Despite these two potential sources of error, a comparison of the RTK-GPS and LiDAR data collected during the same season produced similar enough results that we felt comfortable combining data sets for the analysis.

in 2019 (Kruskal-Wallis chi-squared = 1243.8, df = 269, $p < 0.001$; Figure 14B and Table 8). The difference in median elevations between placement sites in 2019 and 2015 was 0.33 feet by 2019 (Kruskal-Wallis chi-squared = 5084.1, df = 259, $p < 0.001$; Figure 14C and Table 8).

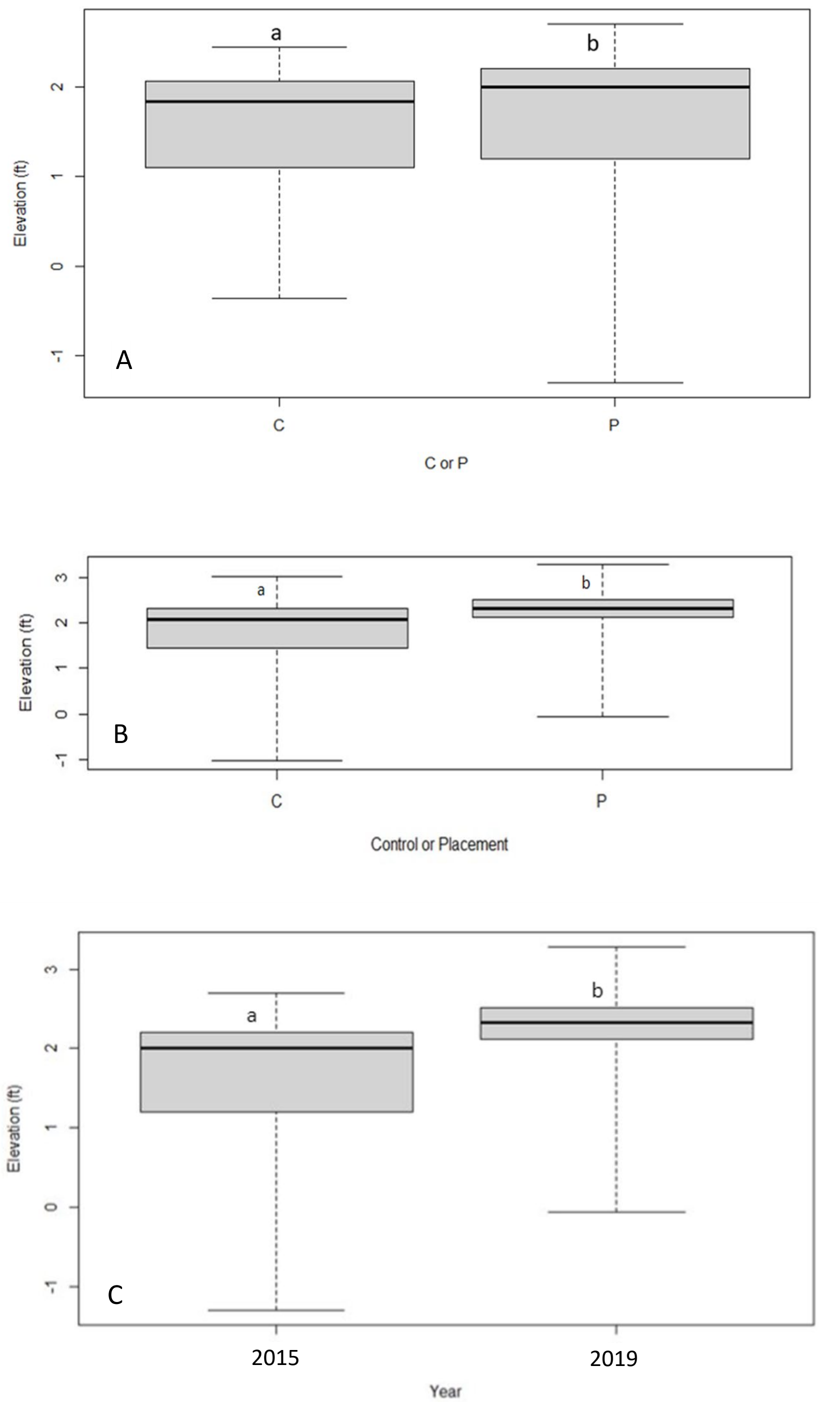
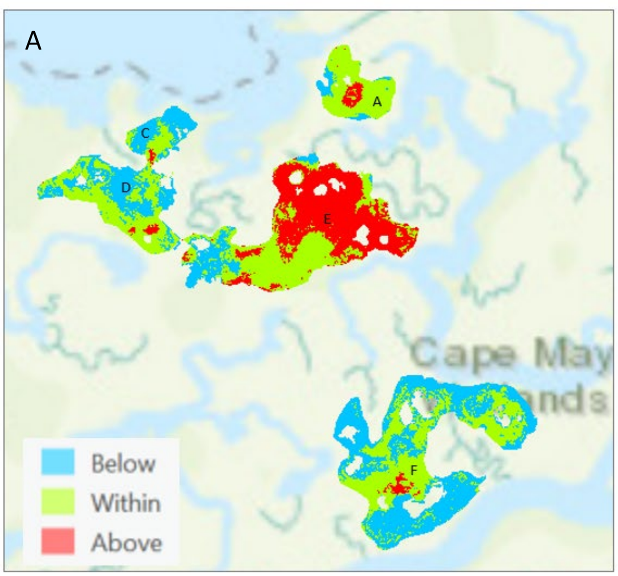


Figure 14. *Box and whisker plots include the minimum, quartiles, median, and maximum elevation of A) baseline conditions (2015) between Control (C) and Placement Areas (P), B) four years post-placement (2019) conditions between Control and Placement Areas, and C) change between baseline conditions (2015) and four years post-placement conditions (2019) in only Placement Areas in Avalon.*

Table 8. *Elevation (Feet, NAVD88) quartiles at Avalon in Control and Placement sites during baseline (2014/2015) and four years post sediment addition in 2019 based on RTK-GPS surveys.*

	Elevation Quartiles (Feet, NAVD88)					
	n	**0%**	**25%**	**50%**	**75%**	**100%**
Control Baseline	66	-0.36	1.12	1.83	2.06	2.44
Control 2019	1434	-1.04	1.44	2.06	2.31	2.61
Placement Baseline	2750	-1.3	1.2	2.0	2.2	2.7
Placement 2019	3162	-0.07	2.12	2.33	2.51	3.28

Avalon Elevation vs Target Elevations: Based on interpolated ground-based LiDAR points, measured mean “as built” elevations were 0.1-0.34 feet below the Target Dredged Material Placement Elevation for all areas except E, which was 0.17 feet above. Given the reported minimum and maximum “as built” elevations, it is clear that some portions of each placement area did not achieve the target placement elevation, while the target elevation was exceeded in other parts of each placement area (in some cases by more than a foot). The “as built” surveys demonstrated that 44% of the area that received sediment was within the target sediment placement elevation, 36% fell below, and 20% was above (Figure 15A). Most of the area above the target sediment placement elevation was in area E where the target elevations were lower than the other areas. By 2019, approximately 50% of the areas were within their Ecological Target Elevation ranges with 30% below and 20% above (Figure 15B).



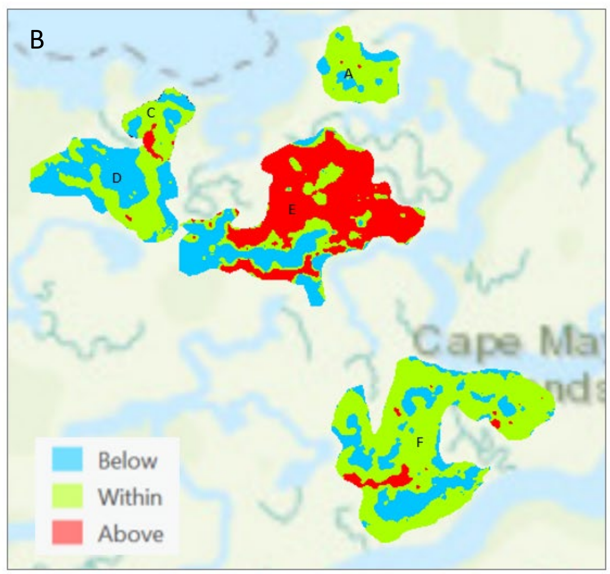


Figure 15. *A) 2016 (“as built”) map of Avalon placement areas that were below, within, or above Target Placement Elevations. Transparent areas have no data because they were ponded. B) 2019 map of placement areas that were below, within, or above Target Ecological Elevations. Each Target Elevation was buffered by 0.25 ft above and below to create a range.*

Fortescue

In Fortescue, the range for low marsh was determined to be -0.39 to 2.66 feet NADV88, and high marsh to be 2.66 to 3.08 feet NAVD88 (see Section 4: Water Level Monitoring). The Fortescue marsh enhancement project was designed to increase the elevation of 23 acres of ditched and low-lying salt marsh to 2.80 to 3.00 feet NAVD88 (in addition to the enhancement of 1,350 linear feet of coastal dune, and the restoration of 3.90 acres of beach; not monitored). Topographic surveys were completed at Fortescue using RTK-GPS between 2015 and 2019.

Fortescue Elevation

Topographic surveys at Fortescue were difficult to draw conclusions from. Median elevations in the placement areas were higher than in the control site before sediment placement (Table 9). Elevation gains in the placement areas were observed after placement but were no longer measurably different from baseline conditions by 2019 (i.e., the difference in the median elevation was below the 0.16-foot accuracy of the RTK-GPS). Another complicating factor was that control sites gained improbable elevation during the 4-year time frame (0.36 feet).

Table 9. *Fortescue elevations over time (Feet NAVD88). Minimum, 25 percent, median, 75 percent, and maximum.*

	Elevation Quartiles (Feet, NAVD88)					
	n	**0%**	**25%**	**50%**	**75%**	**100%**
Control Baseline	1780	-0.90	2.10	2.40	2.70	3.60
Control 2019	1719	0.01	2.46	2.76	3.06	3.96
Placement Baseline	664	-0.08	2.25	2.60	2.79	3.60
Placement 2019	624	1.24	2.62	2.80	2.95	3.59

Fortescue Elevation vs Target Elevations

At Fortescue, the 2016 “as built” surveys demonstrated that 56% of the site was within target elevation ranges, with the majority of the rest of the area below targets (Figure 16A). By 2019, the median elevation in placement areas was on target at 2.80 feet NAVD88 but was uneven across the site (Figure 17). Median elevations were higher in placement areas than control areas in baseline conditions (2015, Kruskal-Wallis chi-squared = 2407.8, df = 745, $p < 0.001$) and in 2019 (Kruskal-Wallis chi-squared = 1253.8, df = 768, $p < 0.001$). Elevations were additionally higher in placement areas in 2019 than during baseline. (Kruskal-Wallis chi-squared = 1253.8, df = 768, $p < 0.001$). However, median elevations in control areas also increased by 0.36 feet between 2015 and 2019 (Kruskal-Wallis chi-squared = 2911.4, df = 325, $p < 0.001$). Despite these increases, the area within the target elevation ranges had reduced to 37%, and the area below the target elevation ranges increased to 53% (Figure 16B).

In addition, while surveys of placement areas and control sites showed gains in elevation from 2015 to 2017, the marsh surrounding sediment placement show a loss in elevation (Figure 18). It is unknown how much of these unexpected results can be attributed to a larger-than-expected error in the surveys and how much of the results are accurate. Elevation loss around placement areas could be the result of compaction during construction activities (like heavy equipment use) or the area being especially vulnerable to sea-level rise.

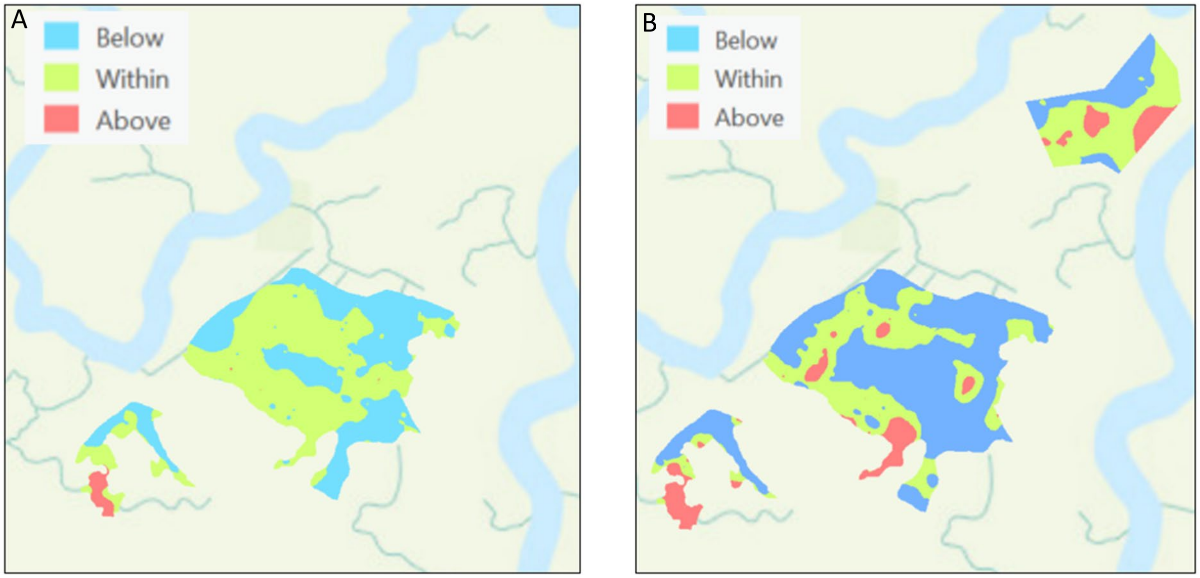


Figure 16. *Fortescue: A) 2016 ("as built") map of areas that were below, within, or above Target Placement Elevations, and B) 2019 map of areas that were below, within, or above Target Ecological Elevations. Each Target Elevation was buffered by 0.25 ft above and below to create a range.*

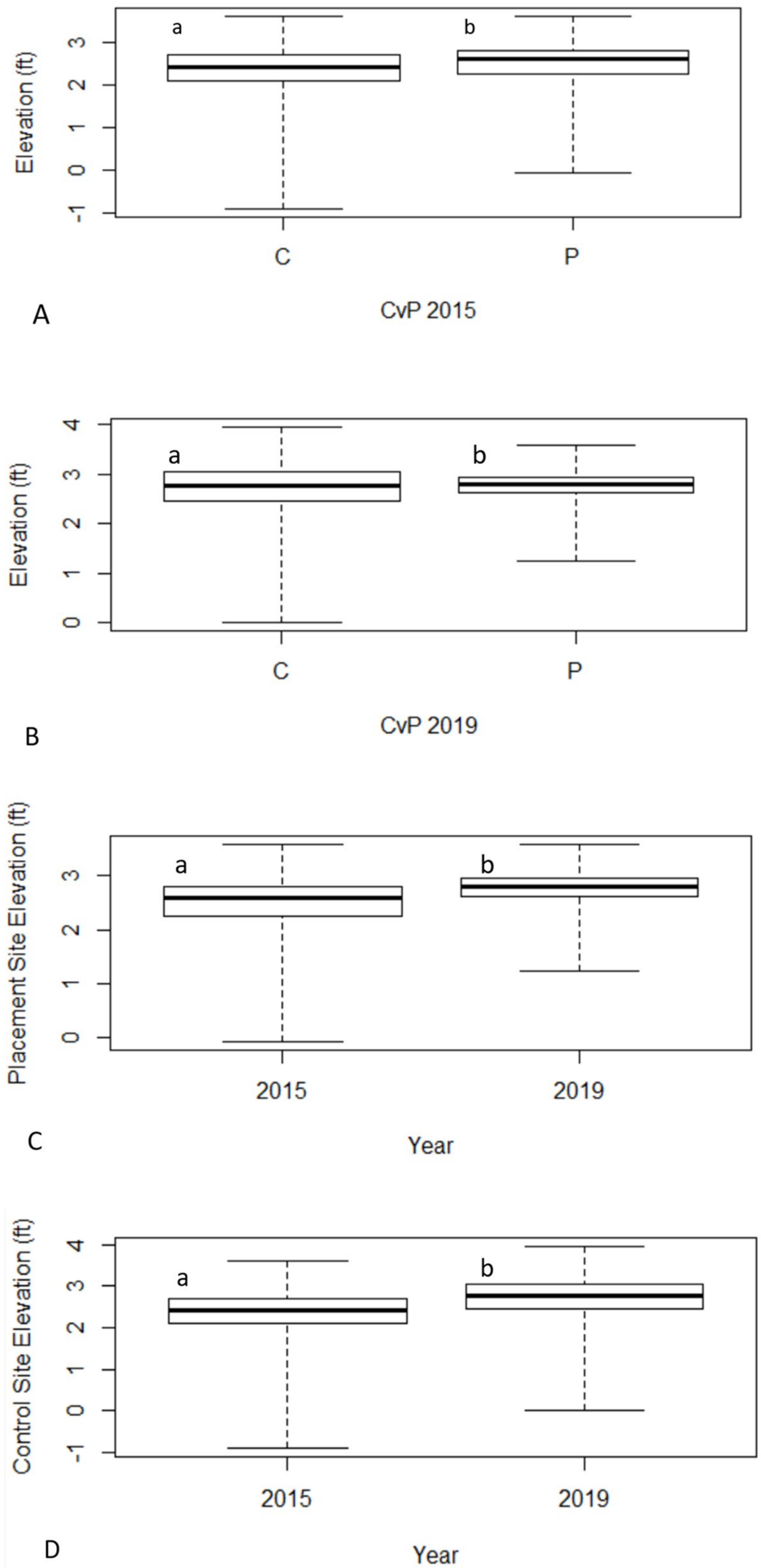


Figure 17. *Median, minimum, maximum and quartile elevations (ft, NAVD88) at Fortescue based on topographic surveys during A) 2015 and B) 2019, with additional comparisons of elevations of C) Placement Areas (P) and D) Control Areas (C) over time.*

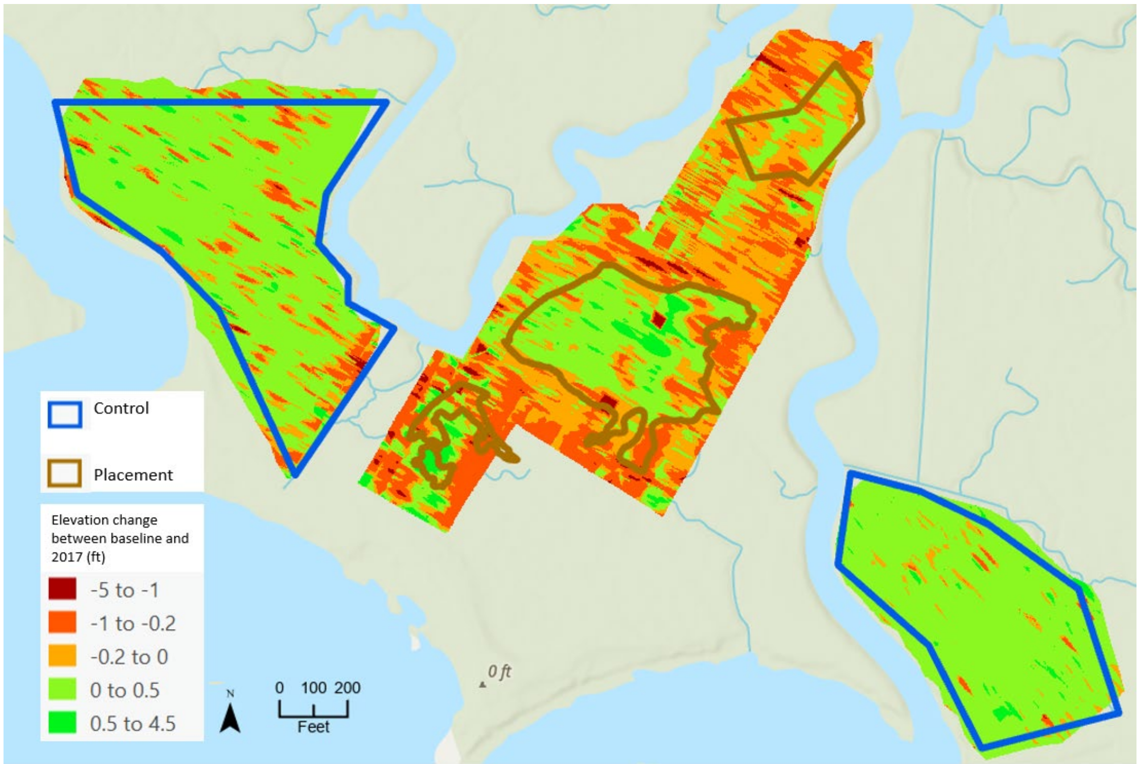


Figure 18. *Change in elevation between 2015 (baseline) and 2017 at Fortescue in Control and Placement areas.*

Conclusions

Elevation

- The relationship between elevation and vegetation is discussed in Section 5: Vegetation from Plot Based Monitoring.
- The dredged material was generally placed at the Target Dredged Material Placement Elevations in around 50% of each site. The remainder of the site was generally higher than planned at Ring Island, lower than planned at Fortescue, and both above and below at Avalon.
- In 2019, Ring Island had maintained the initial elevation gain (median increased by 0.55 feet), Avalon had maintained some of the initial elevation gains (median increased by 0.34 feet), and an elevation change could no longer be reliably measured at Fortescue (median increase around 0.2 feet).
- The current rate of sea level rise in New Jersey is 6 mm/year (Haaf et al. 2016), which is equivalent to 0.02 feet. Given the median elevation gains at the sites, this suggests that Ring Island, Avalon, and Fortescue gained 27.5, 16.5, and 10 years, respectively, before they are back to starting conditions relative to sea level (Table 10).

Table 10. *Elevation gains from placement of dredged material vs years of resilience against the current rate of relative sea-level rise in New Jersey.*

Site	Elevation gains as of 2019 (ft)	Equivalent number of years of SLR
Ring Island	0.55	27.5
Avalon	0.33	16.5
Fortescue	0.20	10

- The Project Team expected to see decreases in elevation over time as the dredged material dewatered, consolidated, and was redistributed within and outside the placement areas as a result of daily tides, storms, and wind. Subsidence of the marsh platform under the weight of the dredged material was also considered a possible contributor to elevation loss. This was especially likely at Avalon, where most of the site began as pools, many with unconsolidated bottoms. Loss of some initial elevation gains was observed at Avalon and Fortescue. Some redistribution of sandy sediments and/or compaction of the marsh in the thin layer placement areas of Ring Island were observed. The ENH at Ring Island also lost elevation as sand was redistributed or compacted the marsh surface. It is possible that because the sediment was coarse, it redistributed around the site rather than completely eroding (for example, a beach formed directly to the east of the original ENH footprint).
- The nesting platform experienced numerous high-water events but was not overtopped after construction. Elevations at constructed nesting habitats will need to be maintained over time to maintain suitable habitat for coastal nesting birds. Suitable nesting habitat is early successional habitat, so poorly vegetated, sandy habitats are ideal. Periodic elevation refurbishment would help maintain these conditions. Sediment redistribution elsewhere on the marsh platform may further enhance adjacent marsh areas by providing additional sediment influx and increasing marsh accretion.
- Despite most areas of Avalon being close to their Target Ecological Elevations, achieving the Target Ecological Elevations and even application of dredged material are likely to be major challenges for sediment addition projects that do not use machinery to contour the dredged material after it has been placed. This results from the difficulty in predicting consolidation of the sediment and subsidence of the underlying marsh platform to set Target Placement Elevations as well as difficulty directing and spreading sediment as it is being placed.
- Based on these results, we can answer the following monitoring questions: 1) Were Target Ecological Elevations appropriate for the creation of native high marsh plant communities, and 2) How much does the placed dredged material consolidate over time?
    - 1) All projects stated the desire to increase marsh elevation to high marsh, targeting primarily *Spartina patens*. Although Avalon and Fortescue had large areas in the target tidal elevation (between Mean High Water (MHW) and Mean Higher High Water (MHHW) levels) for *Spartina patens* and the species was planted at the sites, it did not become the dominant species. Ring Island did not have specific Target Ecological Elevations but instead target thicknesses as this was an initial attempt at placing dredged material and

determining the methodology. On average, we were able to achieve target thicknesses, although some areas exceeded the target range. This sediment placement resulted in elevations above the high marsh zone (MHW to MHHW). The Target Ecological Elevations for Avalon were mixed in terms of appropriate ranges. The Avalon platform was a mixture of low marsh, high marsh, and nearly upland elevations. This was largely due to the inconsistency of placement due to the difficulty of directing and spreading sediments. Fortescue was determined to be largely within high marsh elevations several years after placement, demonstrating that the Target Ecological Elevation was appropriate.

   - 2) Ring Island maintained the initial elevation gain (median increased by 0.55 feet). While some areas of Avalon compacted, particularly around the ponded habitats, there was an overall elevation gain of 0.33 feet NAVD88, suggesting that sediments were subject to horizontal movement rather than vertical. Fortescue demonstrated similar findings, with an estimated elevation gain of 0.2 feet NAVD88 over time, despite some areas compacting.

Monitoring Methods

- Accurate and consistent topographic surveys were a major challenge for this project. RTK survey error was expected to be up to 0.16 feet and there was an additional interpolation error when ArcPro was used to make Digital Elevation Models of the sites. Thin-layer placement projects targeting under 0.5 feet of elevation gain are likely to have a difficult time documenting elevation change using current survey methods.
- There were differences in the topographic survey methods used at Ring Island and Avalon, making comparisons between years at the placement areas difficult, especially at Ring Island. When possible, it would be advisable to use the same survey methods over time. In addition, when using LiDAR data, it must be corrected for vegetation and standing water.
- SETs and marker horizons were installed after sediment was placed on the marsh primarily because of the uncertainty as to what areas would receive sediment, but also because of tight dredging timeframes and concerns about damaging the SETs during placement. This was less than ideal, as the SETs and marker horizons could have been used to look at dredged material consolidation and subsurface compaction of the marsh. In addition, the marker horizons eroded at some sites because they were placed on top of dredged sediments that hadn't stabilized. Plastic grids were installed with new feldspar marker horizons to allow comparison of the two methods.
- At Avalon, it would have been advisable to install permanent platforms to install and monitor the SETs because the unconsolidated, fine-grained sediments were easily disturbed.

4. Water Level Monitoring

Metthea Yepsen[1], GreenVest, Brittany Wilburn[1,2]
[1]The New Jersey Department of Environmental Protection, [2]Drexel University

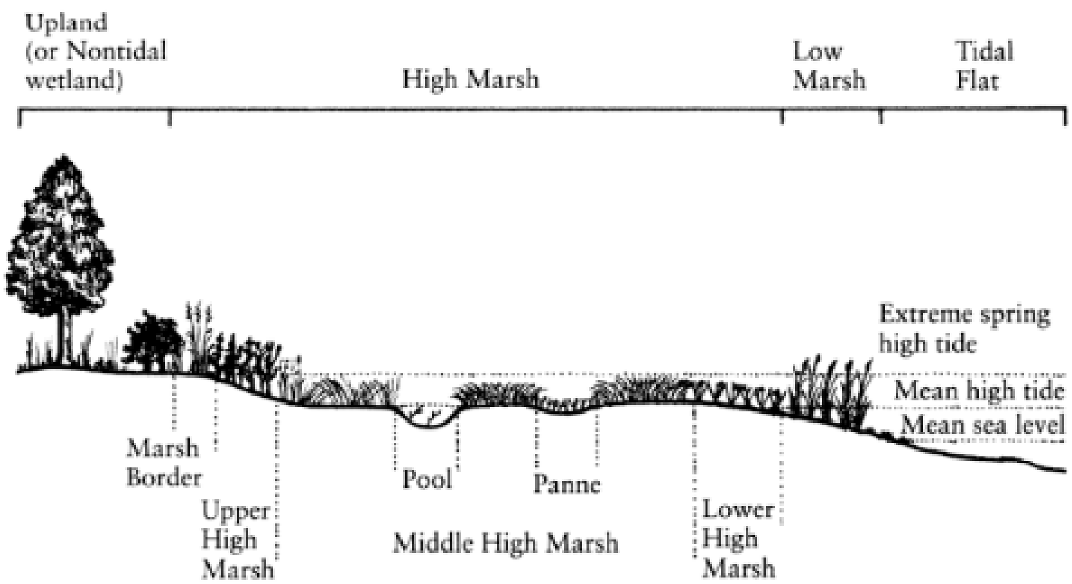


Figure 19. *"Generalized plant zonation in northeastern salt marshes: (1) low mash and (2) high marsh. The high mash can be farther subdivided into several subzones. Pools and depressions called "pannes" occur within the high mash." From Tiner 2009.*

Surface elevation and marsh topography determine the depth and duration of tidal flooding at a site. Tidal flooding and other factors determine what vegetation can grow (Figure 19). All three enhancement projects were targeting high marsh, which generally occurs between Mean High Water and Mean Higher High Water. Since the placement of dredged material changes the topography and elevation of a marsh, it is crucial to monitor changes to the tidal flooding regime (i.e., marsh hydrology). Water surface elevations were continuously measured at Avalon and Fortescue, converted into NAVD88 elevations, and correlated with the site topographic surveys (Table 11). Tidal datums (e.g., elevations of mean high tide, mean tide, etc.) and hydroperiod were also calculated using the water level monitoring data using the available National Tidal Datum Epoch (1983-2001). To account for sea-level rise, water surface elevations were monitored a second time in 2019 in Avalon, Fortescue, and Ring Island in Middle Township. 2019 data was then used to calculate a new tidal datum based on the 2002-2021 epoch following the NJ Tidal Wetlands Monitoring Network (NJTMWN) Tidal Datum Calculation SOP[10].

[10] A draft version can be requested from the NJ Tidal Wetland Monitoring Network by emailing DSR.wetlands@dep.nj.gov

Table 11. *Design and metrics used for surface water level monitoring at the Avalon and Fortescue project sites.*

Design	Hydrology
B-A-C-I	Hydroperiod
Continuous monitoring from February 2015 to December 2017 Additional continuous monitoring from June to October or November 2019	Mean Higher High Water (MHHW) Elevation
	Mean High Water (MHW) Elevation
	Mean Tide Level (MTL) Elevation
	Mean Low Water (MLW) Elevation
	Mean Lower Low Water (MLLW) Elevation

Monitoring Design

Surface water elevations were continuously measured by GreenVest at Avalon and Fortescue from February 2015 through early-December 2017. Measurements include water elevations during baseline data collection before dredged material placement, during placement, and throughout the monitoring period after placement. Remote data logging equipment was installed at semi-permanent tide gauge stations at the Avalon and Fortescue project sites. This BACI design was intended to provide the data needed for important analyses of the relationships between changes to elevation and tidal flooding caused by dredged material placement and changes to other marsh features (e.g., plant community changes as a result of differential flooding regimes).

In situ Level Troll 500 water level loggers were installed in slotted PVC pipe in open water, in a creek, in a control site, and in two placement (impact) locations at both Avalon and Fortescue. Water level elevation was measured at 15-minute intervals. Periods of anomalies and logger malfunctions were removed from the dataset before analysis. Key tidal elevations (Table 11) were calculated, when possible, and validated by Princeton Hydro using long-term data from NOAA tide gauges.

In 2019 surface water elevations were continuously measured by NJDEP from June to October (Fortescue) or November (Ring Island and Avalon). Semi-permanent tide gauge stations were positioned near existing SETs in each site, with two stations in the marsh interior and one station in a creek bed. Tide gauge stations were formed using a slotted PVC pipe with a HOBO U20L-04 water level logger inserted and a second HOBO U20L-04 logger acting as a barometric logger placed nearby. Water elevation levels were collected at 6-minute intervals to coordinate with NOAA tide gauges and to make later calculations easier. Data were checked for errors and corrected or removed when needed. The same key tidal elevations as above were calculated and checked against NOAA tide gauges, using the methods found in the NJTWMN Tidal Datum Calculation SOP. In brief, high and low tides of the short-term monitored water level data were parsed out and used to determine key tidal datum elevations. These elevations were then corrected relative to long-term tidal datum elevations of permanent tidal gauges of NOAA or the USGS, depending on the site. The tidal datums calculated using 2019 data were found to be similar to the design tidal datums of Fortescue and Avalon and are reported here as a secondary confirmation of design calculations.

Results

Ring Island

For the Ring Island TLP areas, the design tidal datum was calculated to have a maximum tidal elevation of

2.14 ft (Table 12). Secondary 2019 calculations were shown to have a relative percent difference of only -1.14%, suggesting that the original tidal datum calculations are accurate. No other monitoring of the surface water elevations was performed at this site.

Table 12. *Tidal datum boundaries calculated for Ring Island. The 2019 datums are based on water level monitoring conducted in a tidal creek at the projects site using the 2002-2021 National Tidal Datum Epoch. Median placement area elevations based on RTK-GPS point surveys.*

(Feet NAVD88)	Tidal Datums Elevation			Median Placement Area Elevation	
	NOAA Stone Harbor Tidal Gauge 8535581	Design	2019	2014 Baseline	2019
Mean Higher High Water	2.08	2.14	2.11		2.50
Mean High Water	1.69	-	1.72	1.95	
Mean Sea Level	-0.32	0.23	-		
Mean Tide Level	-0.40	-	-0.35		
Mean Low Water	-2.50	-	-2.41		
Mean Lower Low Water	-2.66	-2.41	-2.55		

Before placement, half of Ring Island was between Mean Tide Level and Mean High Water and half was between Mean High Water and Mean Higher High Water (Figure 20). In both “as built” (not shown) and year 5 post-construction surveys, elevations at Ring Island had increased and surpassed the tidal datum elevations calculated for high marsh habitat and were mostly above Mean Higher High Water (Figure 20).

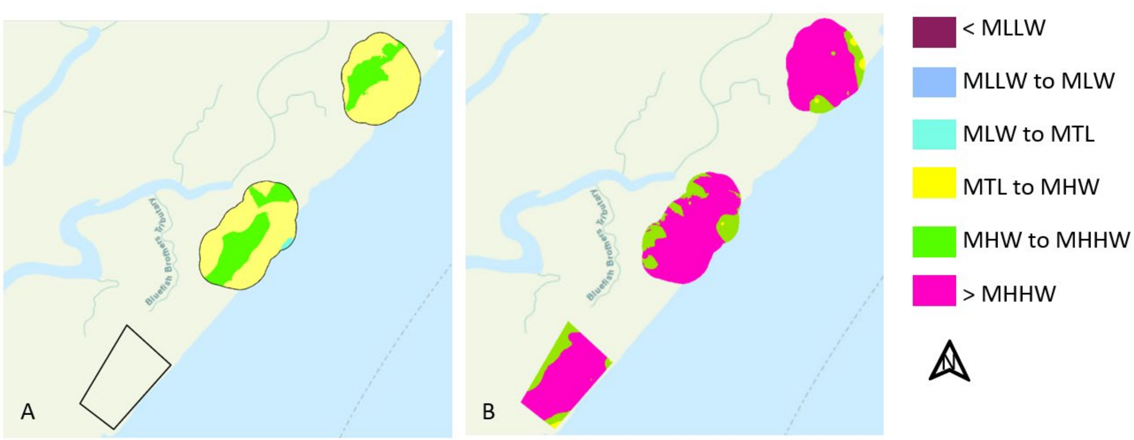


Figure 20*. Elevations relative to tidal datums at Ring Island A) Interpolated RTK-RPS survey from 2014, just before sediment placement. B) Interpolated RTK-GPS survey from 2019. Tidal datums were calculated using water level data collected in 2019.*

This is contrary to what was expected, as it was assumed that sediments would compact and reconsolidate after placement.

No target elevations were set for Ring Island. Thus, we cannot compare them to tidal datums.

Avalon Phase 2

GreenVest installed tide gauges on the site to observe tidal inputs and fluctuations and to monitor inundation, saturation, and drainage of the marsh plain. The tidal datums that were used for design were developed using two months of data compared to other gauging stations for reference. Tide elevations were obtained from National Oceanic & Atmospheric Administration (NOAA) VDatum program and tide gauges in Atlantic City (Station ID: 8534720; Table 13). However, subsequent monitoring showed a larger difference in surface water elevations for MHHW, with a relative percent difference of -8.73% in 2016. The secondary 2019 calculations were referenced to Stone Harbor NOAA tide gauges, and results were similar to 2016 monitoring levels and had a relative percent difference of -10.11% compared to the design MHHW levels.

Table 13 compares the tidal datums to the Target Ecological Elevations, the median baseline elevations, and the median elevations in 2019. The median placement elevation in Avalon placement areas started between Mean Tide Level and Mean High Water at the upper end of the low marsh range that was used for design. Target Ecological Elevations for Areas A, D, and F were all above the Mean Higher High Water elevations used during design. The Target Ecological Elevation for Area C was between Mean High Water

Table 13. *Tidal datums delineated from tide gauge MW-2 in 2015 and 2016 at Avalon and tide gauge located in a creek in 2019; compared with datums used for project design. 2017 data was not usable due to logger malfunctions. Mean Water level from 2015 was calculated using the 2015 Avalon data set at MW-2. Median placement area elevations based on RTK-GPS point surveys.*

Elevation (Feet NAVD88)	Tidal Datums						Target Ecological Elevations	Median Placement Area Elevation	
	NOAA Stone Harbor Tidal Gauge 8535581	NOAA Atlantic City Tidal Gauge 8534720	Design	2015	2016	2019		2014/15 Baseline	2019
Mean Higher High Water	2.08	2.39	2.39	2.35	2.19	2.16	2.5 (Areas A, D, F)		
Mean High Water	1.69	1.97	2.03	2.01	1.83	1.79	2.1 (Area C)		2.33
Mean Tide Level	-0.40	-0.03	-	-0.03	-	-0.34	1.9 (Area E)	2.0	
Mean Low Water	-2.50	-2.04	-2.00	-2.29	-2.49	-2.47			
Mean Lower Low Water	-2.66	-2.21	-2.61	-2.46	-2.79	-2.64			

and Mean Higher High Water, and the Target Ecological Elevation for Area E was between Mean Tide Level and Mean High Water. In 2019, four years after sediment was added, the median elevation of placement areas at Avalon was at the upper end of the Mean High Water to Mean Higher High Water range in the high marsh.

Figure 21 shows the spatial distribution of elevation ranges at Avalon parsed into tidal datums. During baseline conditions, the majority of both placement areas and controls were in the low marsh range between Mean Tide Level and Mean Higher High Water. After additions of sediment, followed by consolidation and subsidence, a wide elevational gradient from Mean Tide Level in yellow to above Mean Higher High Water in pink was observed. Interpretation of the water level data is complicated by the pools at Avalon that hold water as tides recede and do not receive tidal water until their bank elevations are

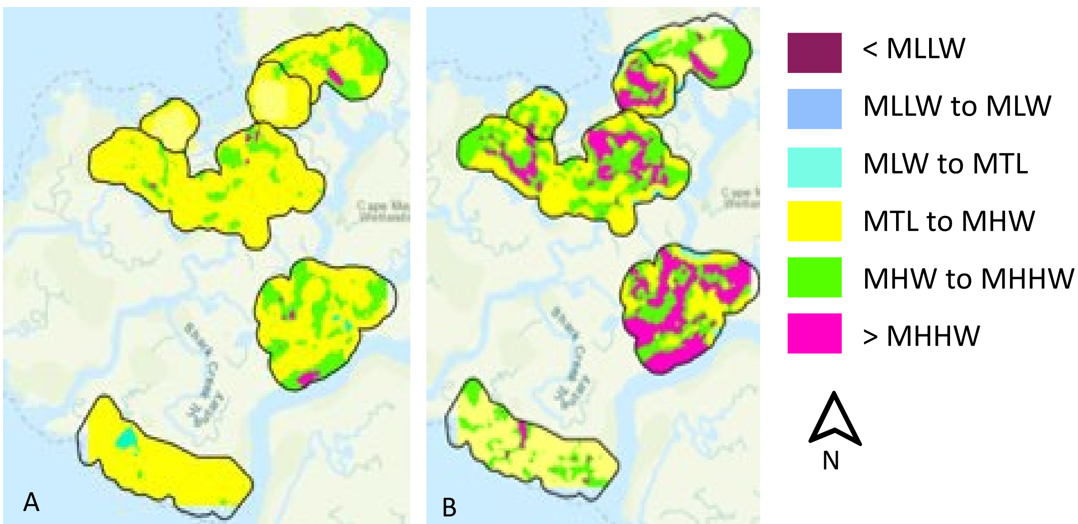


Figure 21. *Tidal datum elevations of Avalon in A) baseline (2014/15) and B) four years post-construction in 2019. Tidal datums are those used for project design and were based on water level data collected at the site in 2015 and are comparable to those used to set target elevations.*

exceeded. This made it difficult to determine the percentage of time an area of the Avalon marsh was flooded since it cannot be based on elevation alone.

Fortescue

Tidal datums used to set target elevations at Fortescue were based on three months of data collected at the site. Because the onsite data was from a relatively short period, tide range data was compared to other gauging stations for reference. Tide elevations were obtained from NJDEP Office of Engineering and Construction Bureau of Coastal Engineering Project 2155, page 2 of "Proposed Emergency Dredging Fortescue Creek, Township of Downe, County of Cumberland, dated June 12, 2013. These elevations were converted to NAVD88 using the NOAA Vdatum datum transformation program. Datums recorded from

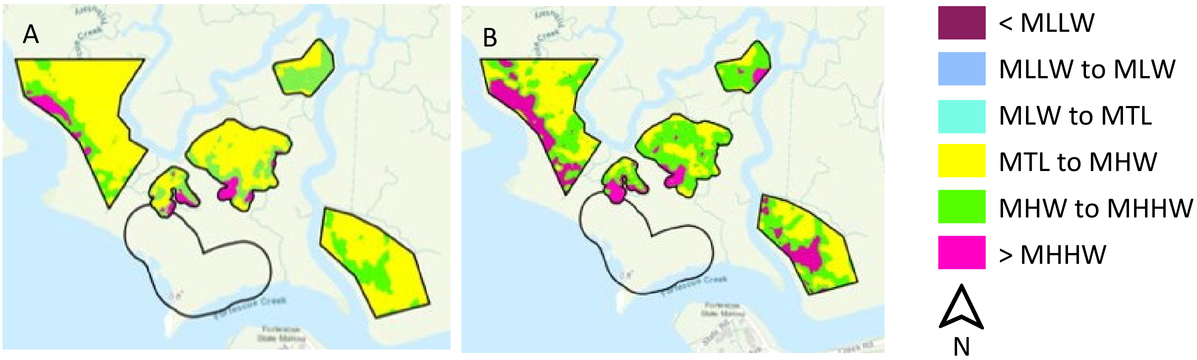


Figure 22. *Tidal datum elevations of Fortescue in A) baseline conditions in 2015 and B) four years post-construction in 2019. Tidal datums were those used in project design.*

2015 to 2019 in Fortescue Creek near the project site (Figure 22) were a few inches lower than those used to set the Target Ecological Elevations for the marsh enhancement project (Table 14).

Figure 22 compares the baseline hydrologic conditions at Fortescue in 2015 (A) to the conditions two years after dredged material placement (2019; B). For the analysis, tidal datums were calculated using the NOAA Vdatum transformation program on tide elevations from the NJDEP Office of Engineering and Construction Bureau of Coastal Engineering Project 2155. These tidal datums were used to design Fortescue placement depths. In the areas that received sediment, some locations (mostly in Area 2) have increased in elevation from between MTL and MHW to between MHW and MHHW.

Table 14. *Tidal datums calculated based on tide gauge TG-1 in 2015 and 2016 and on quality check tide gauges at Fortescue in 2019 compared to the tidal datums used for project design. Median elevations in placement areas calculated from RTK-GPS point survey.*

Elevation (Feet NAVD88)	Tidal Datums					Target Ecological Elevation	Median Elevations in Placement Areas	
	NOAA Bivalve Tidal Gauge 8535055	Design	2015	2016	2019		2015 Baseline	2019
Mean Higher High Water	2.86	3.08	2.81	2.82	2.85			
Mean High Water	2.43	2.66	2.47	2.43	2.45	2.8 to 3.0		2.8
Mean Tide Level	-0.39	-			-0.24		2.6	
Mean Low Water	-3.22	-3.29	NA	-2.37	-2.93			
Mean Lower Low Water	-3.41	-3.47	NA	-2.51	-3.10			

Conclusions

- Based on tidal datums, Ring Island started as an even mix of low marsh (MTL-MHW) elevations but tended toward already being in high marsh elevation (MHW-MHHW); Avalon and Fortescue started at the upper end of low marsh elevation ranges.
- In 2019, four to five years post sediment addition, Ring Island was predominantly above high marsh elevation (>MHHW); Avalon was fairly evenly split between low marsh, high marsh, and above high marsh elevations; and Fortescue was predominantly in the upper end of high marsh elevations.
- As previously noted, elevation data at Fortescue is complicated by the fact that larger-than-expected increases in elevation were measured in the control sites, suggesting that there may be errors in the data.
- One way to use the analysis of tidal datums is to consider whether or not the correct Target Ecological Elevations were used in the project design. For this project, we wanted to see if we could bring the elevation of the marsh to the upper end of the high marsh range. At Ring Island, where elevation targets were not used, we can see that the addition of a median of 0.55 feet of elevation moved the site from low marsh to above high marsh, which was perhaps too high. Raising the elevation above MHHW may explain why the site has been slower than Fortescue and Avalon to revegetate (see Section 5: Vegetation from Plot Based Monitoring). At Avalon, three out of the five placement areas had Target Ecological Elevations above MHHW and anecdotal evidence from site visits has suggested that persistent bare areas tend to correlate with the lowest or highest elevations. Fortescue targets were consistent with the upper end of the high marsh range.
- The range of elevations that fall into the high marsh tidal datums (approximately 0.5 feet) is much smaller than the range of elevations that fall into the low marsh tidal datums (generally 2-3 feet in marshes in SE New Jersey), creating some inherent challenges for using TLP techniques to reach high marsh elevations because the room for error is much smaller.

5. Vegetation from Plot Based Monitoring

Elizabeth Muntean[1], Metthea Yepsen[1,2], Lori Lester[2], Adrianna Zito-Livingston[1], Doug Faircloth[1], Laura Crane[1], Jessie Buckner[1], Jaclyn Woollard[1], Jaqueline Jahn[1]
[1]The Nature Conservancy, [2]New Jersey Department of Environmental Protection

Salt marsh vegetation is the primary driver of the net increase in marsh elevation, through the trapping of sediment from the water column and the build-up of slowly decaying plant material. The re-establishment of vegetation after the placement of dredged material is necessary for project success and will also likely correspond with all other environmental monitoring metrics. Monitoring vegetation provides information about the effects of placing dredged material on a variety of stressed salt marsh vegetation communities. Elevation and tidal flooding are important determinants of vegetative community composition, and the placement of dredged material on the marsh plain will be accompanied by immediate changes in marsh elevation and the depth and duration of flooding. Dredged material composition and thickness may also affect the recovery of vegetation after placement. The goal for monitoring vegetation was to determine the response of vegetative communities to placement, identify trends in vegetative recovery and ecological uplift over time, and identify the factors influencing vegetative recovery.

Monitoring Design

***Table 15.** Design and metrics used for TNC's vegetation monitoring at the Ring Island, Avalon, and Fortescue project sites. *Metrics were collected in a subset of plots only in certain years.*

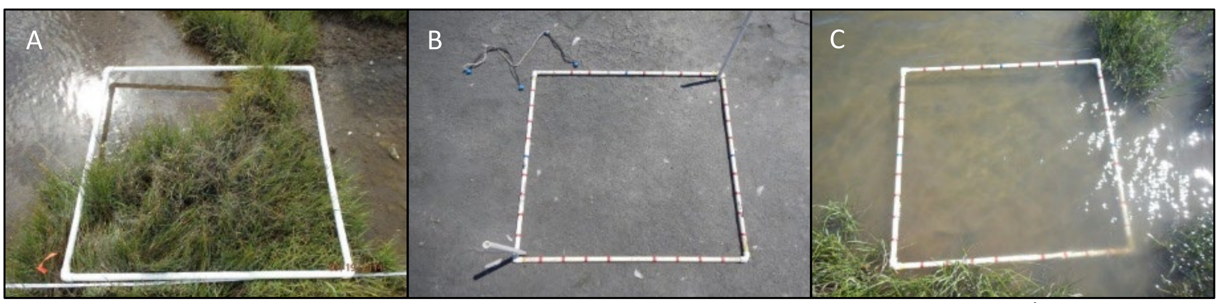
Design	Vegetation Monitoring Metrics
B-A-C-I	Species Richness and Composition
Once Annually During Peak Growing Season (July to September)	Average Stem Height of Dominant Species*
	Percent Cover by Species
	Above-Ground Biomass*
	Below-Ground Biomass*

Vegetation metrics were monitored in permanent 1m² plots annually, except during 2020 (Tables 15 and 16). Plant biomass was monitored in a 0.25m² plot adjacent to a subset of these permanent monitoring plots. Plants were identified to determine species richness, and dominant species were defined as species that covered more than 50% of the plot. Epifaunal macroinvertebrates (ribbed mussels, crabs, snails), sediment depth, and bearing capacity were also monitored within or next to these permanent vegetation plots. Spatial considerations for experimental design were site-specific and included variations in habitat types and elevation so that low marsh, high marsh, marsh plain, and pools were all represented in the data set.

Additionally, each plot was classified as one of three habitat types based on visual observation: if ≥ 50% of the plot was vegetated, it was classified as "vegetated"; if the plot was < 50% vegetated, it was classified as “bare”; if the plot was >50% standing water, it was classified as a “pool” (Figure 23A-C). During construction, some plots did not receive dredged material, and these plots were ultimately excluded from the analyses.

Table 16. *Labels used in this section and the corresponding years they signify relative to placement of dredged material.*

Label	Corresponding Year Referenced		
	Ring Island Placement	**Avalon Placement**	**Fortescue Placement**
PRE	2014	2014/15	2015
PP1	2015	2016	2016
PP2	2016	2017	2017
PP3	2017	2018	2018
PP4	2018	2019	2019
PP5	2019	NA	NA
PP6	NA	2021	2021
PP7	2021	NA	NA



Vegetated Bare Pool

Figure 23. *Photo examples of the classification of quadrats as vegetated, bare, or pool habitats.*

Results

Habitat Proportions

Ring Island

Before placement, the control and placement areas did not have a significantly different proportion of habitats (Figure 24; adjusted Fishers; $p > 0.5$). 75% of the plots in the treatment area converted from vegetated to bare in PP1. In PP7 treatment plots were beginning to show the same proportion of vegetation as control plots (Fishers; $p > 0.05$), with 75% of treatment plots vegetated in the final year. Control plots remained more highly vegetated compared to PP7 treatment plots (Fishers; $p < 0.05$).

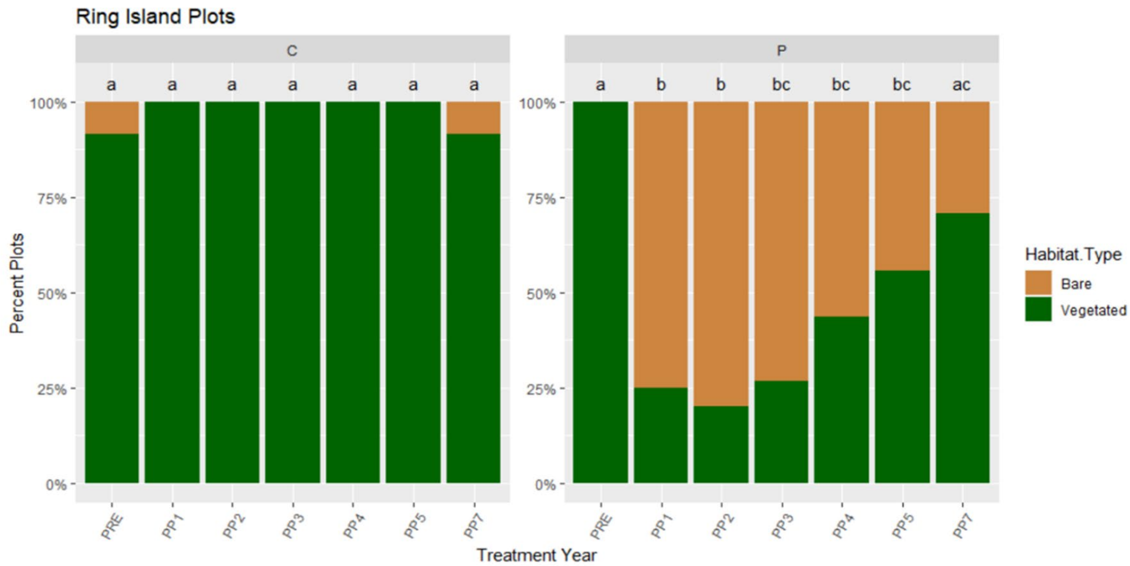


Figure 24. *Proportion of Ring Island monitoring plots categorized as bare or vegetated in control and placement areas pre- and post- placement (n = 12 for controls, n = 16 for placement, alpha = 0.05). Compact letter displays show which groups are not significantly different from one another by two-sample Wilcoxon tests.*

Avalon

Baseline conditions were different between control and treatment areas (Figure 25A; $p < 0.05$). 77% of control plots were vegetated with most of the remaining plots considered to be in pools, compared with 52% vegetated plots and 45% pool plots in the treatment area. The proportion of habitat types in the control plots did not change significantly over time (adjusted Fishers; $p > 0.05$) and changes noted between pool and bare plots have been due to the tidal stage during which the plots were sampled.

After sediment was placed on the marsh, the proportions of habitats changed significantly in treatment plots as both pool and vegetated plots converted to bare areas (Fishers; $p < 0.001$). The cover of this habitat type jumped from 2% to 80%. Over time and with re-vegetation, treatment plots are beginning to look more like control plots (Figure 25B). By PP6, treatment plots had decreased the proportion of pool plots (down to 15%) and increased the proportion of vegetated plots from baseline (up to 65%) but remained below control site proportions (Fishers; $p < 0.05$). Interestingly, the proportion of vegetated treatment plots that started as marsh platform (i.e., not a pool) increased more rapidly than in plots that started as pools (Figure 25C).

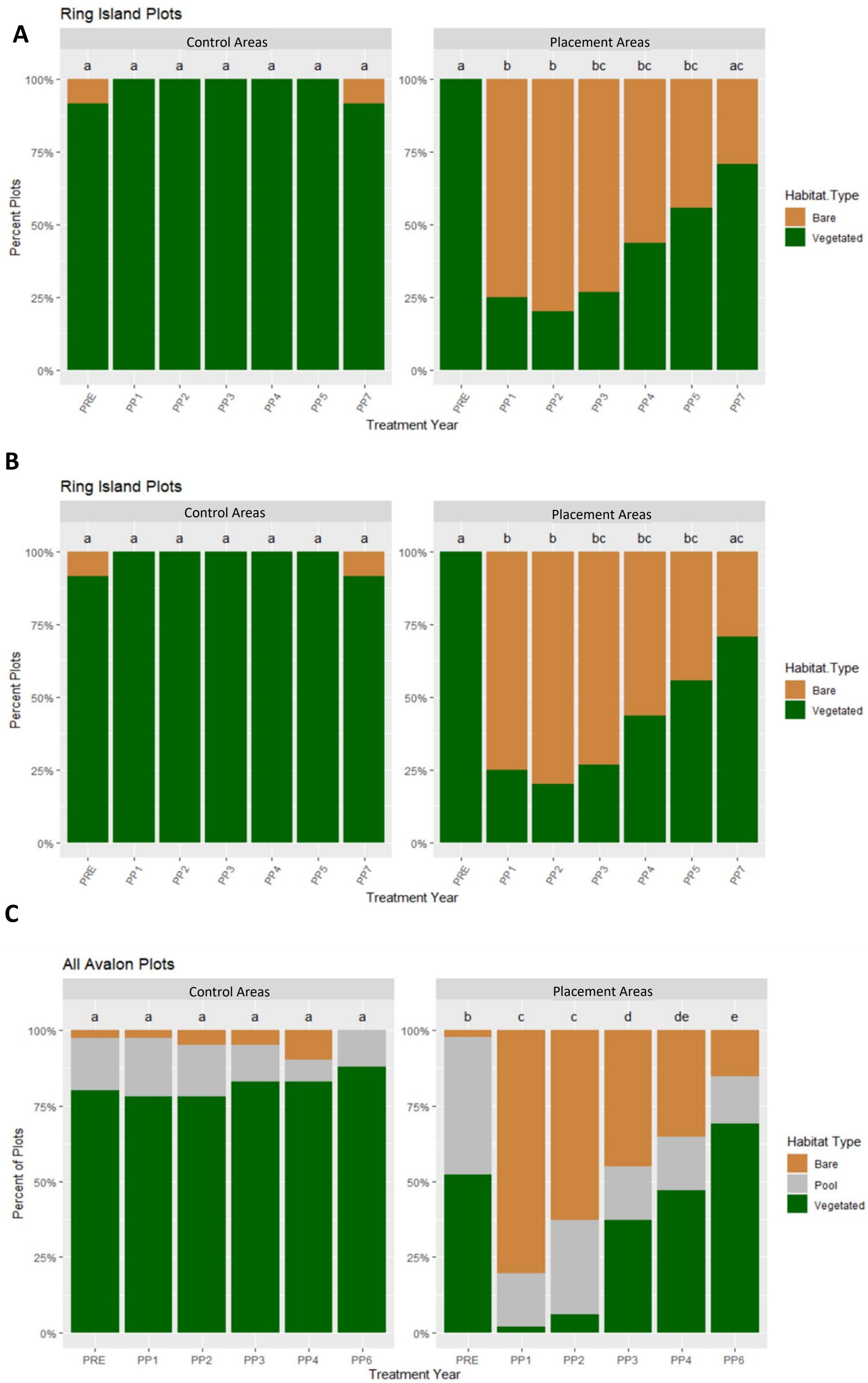


Figure 25. *Proportion of plots at Avalon that were bare, pool, or vegetated in control and placement plots. A) all plots combined, B) plots that started as marsh platform, C) plots that started at pools.*

Fortescue

Prior to placement, the control and placement areas did not have a significantly different proportion of habitats (Figure 26; Fishers; $p > 0.05$), and the proportion of habitat types in the control plots did not change significantly over time (adjusted Fishers; $p > 0.05$). After sediment was placed on the marsh, portions of habitats changed significantly in treatment plots by converting to bare areas or pools (Fishers; $p < 0.001$). Bare habitat type jumped to 84% after placement but then decreased to 0% by the final year. In PP6, 100% of the treatment plots were vegetated, as were 100% of the control plots.

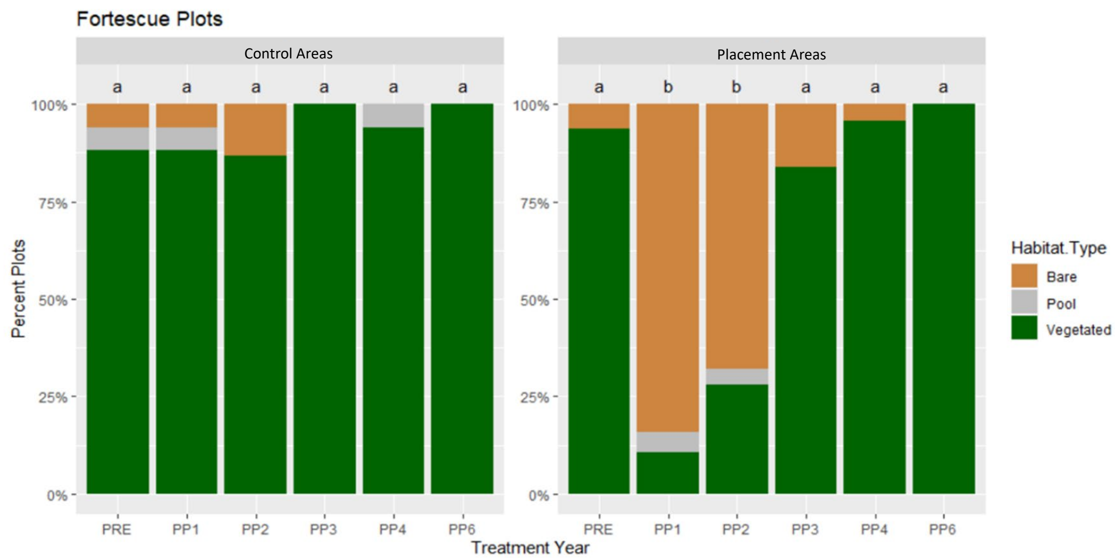


Figure 26. *Proportion of plots at Fortescue that were bare, pool, or vegetated overtime in Placement (P) and Control (C) areas.*

Vegetative Cover

Ring Island

Initial conditions between control and placement plots were not significantly different (Figure 27; Mann Whitney U; $W = 99.5$; $p > 0.05$). Control plots averaged 74% cover and treatment plots averaged 61% cover in the year before placement. Both control and treatment areas experienced a significant change in vegetation cover over time (Friedman $X^2(6) = 39.23$; $p < 0.01$ for controls, $X^2(6) = 17.16$; $p < 0.01$ for placement). In treatment areas, vegetation cover declined one-year post-placement to 20% cover and then showed some recovery in the interim. But, as of PP7, treatment plots average only 8% vegetative cover, significantly lower than baseline conditions (Wilcoxon $Z = -2.43$; adjusted $p < 0.05$). Vegetation cover in control plots significantly declined from initial conditions to an average of 30% cover in PP7 (Wilcoxon $Z = -2.81$, adjusted $p < 0.01$).

Vegetation cover at Ring Island declined immediately after placement. Since then, little vegetation has recovered in monitoring plots and levels remain lower than baseline and control conditions.

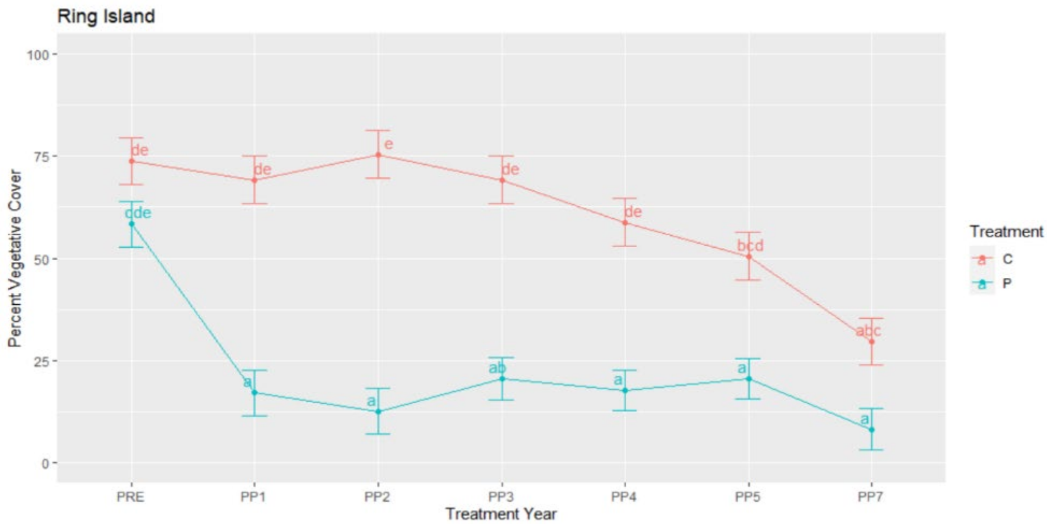


Figure 27. *Percent cover by plants at Ring Island in plots in both Control (C) and Placement Areas (P) overtime. Data presented are means ± one standard error.*

Despite the increasing trend in the proportion of plots that are vegetated, the trend in vegetation cover at Ring Island showed that the density of plants in placement areas stagnated in the seven years since placement Additionally, control site vegetation cover has deteriorated since PP2, and vegetation cover has deteriorated in the treatment plots since PP4.

Avalon

PRE placement and control sites had similar plant cover (Mann Whitney U; W = 1023.5, $p > 0.05$), and both areas experienced a significant change in vegetation cover over time (Friedman X 2(5)= 58.98; $p < 0.001$

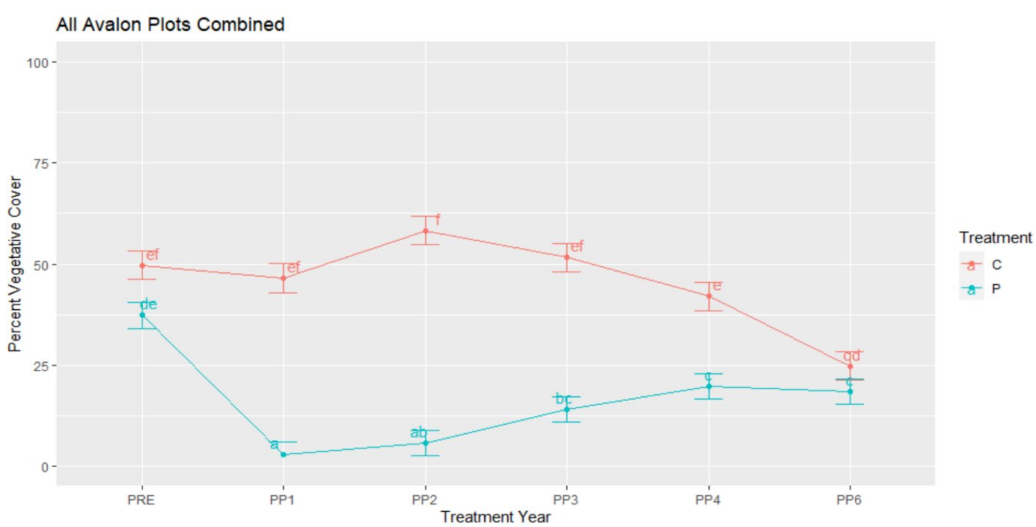


Figure 28. *Percent cover by plant at Avalon in plots in both Control (C) and Placement Areas (P) overtime. Data presented are means ± one standard error.*

for controls; X 2(5)= 64.57; $p < 0.001$ for placement). Large losses in vegetation cover were observed in placement PP1 plots (Wilcoxon Z = -3.9; $p < 0.001$), with vegetation cover dropping from 38% to 3.5% (Figure 28). Six years post-placement, the difference in vegetation cover was not significant between placement plots and control (Mann Whitney U W = 1269.5; $p > 0.05$), with average vegetation cover at 25% and 20%, respectively. Control plot averages, however, are significantly lower than baseline conditions in PP6, down from 50% cover in PRE conditions (Wilcoxon Z = -3.34; $p < 0.001$). Vegetation cover in the placement plots at Avalon declined immediately following sediment placement. After six years of recovery, vegetation cover in placement plots was not different from vegetation cover in control plots and baseline conditions. Overall, these results show large losses in vegetation cover at Avalon in placement areas one-year post-placement followed by recovery in the treatment areas to the same level of vegetation cover as the control areas, though not to baseline, and have not demonstrated uplift.

Fortescue

Placement and control plots did not initially differ in vegetation cover before dredged material was placed on the marsh (Figure 29; Mann Whitney U $W = 133$; $p > 0.05$), though vegetation cover was affected by time in both the placement and control areas (Friedman $X^2(5) = 17.5$; $p < 0.01$ *for controls,* $X^2(5) = 40.5$, $p < 0.0001$ *for treatment)*. Vegetation cover in PP1 plots was significantly reduced in placement areas from 51% to 11% (Wilcoxon $Z = -2.97$; $p < 0.01$).

By PP3, vegetation cover in placement plots was similar to cover in control plots (Mann Whitney U, $p > 0.05$) and was not statistically different from baseline conditions in placement plots (Wilcoxon $Z = -1.14$; $p > 0.05$). Average vegetation cover in both treatment and control plots declined from PP4 to PP6. In placement plots, vegetation cover became significantly lower in year 6 than in initial baseline conditions (Wilcoxon $Z = -2.35$; $p < 0.05$). Control plots declined from 61% to 30% average vegetation cover.

Vegetation cover at Fortescue declined immediately after placement. Three years after placement, vegetation cover was not different from baseline or control conditions. However, the sixth-year

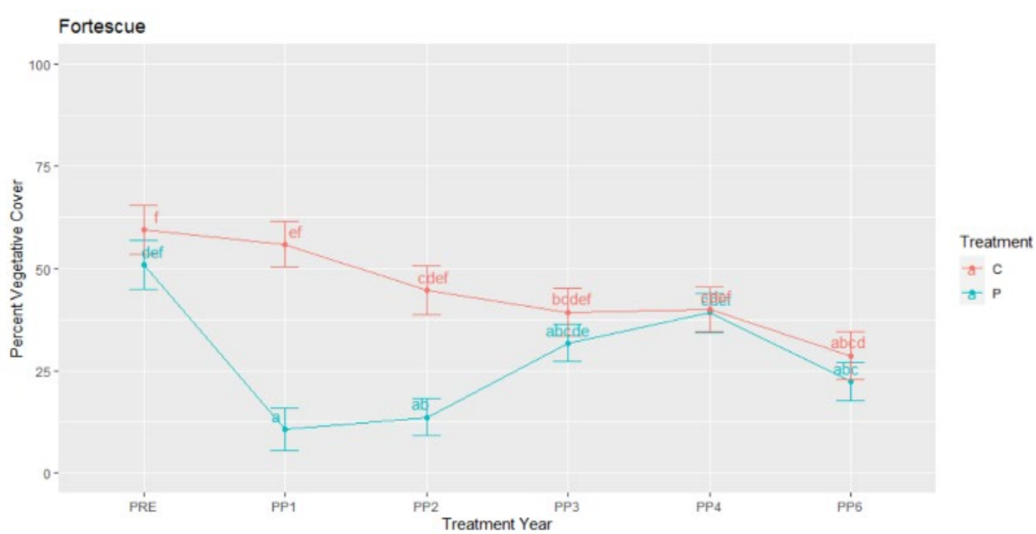


Figure 29. *Percent cover by plants at Fortescue in Control (C) and Placement Areas (P) overtime. Data presented are means ± one standard error.*

vegetation cover in both control and treatment areas had declined. While control and treatment percent cover levels are not significantly different from one another in year six (Mann Whitney U $W = 216$; $p > 0.05$), treatment plots display lower vegetation cover than their baseline conditions.

Overall, these results show that vegetation cover significantly decreased in placement areas one year after placement. Placement plot cover increased in the years after placement, but vegetation cover remains significantly lower than baseline conditions. While placement plot vegetation cover does not differ significantly from current control plot vegetation cover, control plot vegetation cover has slowly declined throughout the study.

Species Richness and Composition

Treatment plots primarily revegetated with *Spartina alterniflora*, a low marsh species, after sediment placement despite our initial goal of facilitating high marsh species habitat (characterized by *Spartina patens*). Richness remained low at all sites in both control and treatment plots in all years with individual plots generally containing one or two species and no more than four, as is typical of salt marsh islands (Niering and Warren 1980).

Despite low levels of native species richness initially, there was a significant decrease in species richness in treatment PP1 plots (Tukey; $p < 0.05$) at all three sites. Treatment plots at all sites regained pre-placement levels of native species richness within the timeframe of the study. Avalon platform and pool plots attained former species richness in PP3 (Tukey; platform $p > 0.05$, pool $p > 0.05$), Fortescue attained baseline species richness by PP2 (Tukey; $p > 0.05$), and Ring Island achieved it in PP3 (Tukey; $p > 0.05$).

Examination of vegetation species composition across all sites is another important measure of restoration success. Notable shifts were observed in native vegetation species at Avalon as plots re-established and became fully vegetated, as *Salicornia spp.* And *Distichlis spicata* briefly dominated some plots before *S. alterniflora* was established. *Phragmites australis* appeared in two study plots at Fortescue nearest to a sand dune restoration project where *P. australis* existed before restoration. The two plots that are dominated by *P. australis* are directly adjacent to the dunes, and there is no sign of *P. australis* spreading to the interior of the marsh as of PP6. Ring Island saw little vegetation recovery and, therefore, little change in species composition except for one *D. spicata*-dominated plot in the third year post-placement.

Ring Island

A mixed effects ANOVA showed that vegetative species richness was significantly affected by the interaction between treatment and time (Figure 30; $p < 0.001$), although changes were small. Before placement species, richness was not significantly different between control and placement areas (Tukey; $p > 0.05$). Within the placement area, species richness significantly declined post-placement (Tukey; $p < 0.01$) from an average of fewer than two species per plot to an average of less than one per plot. Since PP3, species richness has not been significantly different from baseline (PRE) condition, and since PP4 treatment plot richness has not varied significantly from control plot richness (Tukey; $p > 0.05$). Within the control area, there has been no significant change in richness over the seven years. Overall, these results show a significant decrease in Ring Island species richness in placement areas from PRE to PP1, with an increase each year since placement, however, richness remained low in both areas in all years, with individual plots containing no more than three species.

The composition of dominant species in Ring Island plots did not shift after placement in treatment plots. *Spartina alterniflora* was the only dominant vegetation in control plots and was the only dominant vegetation in treatment plots except for one plot in PP3 that was dominated by *Distichlis spicata* (Figure 31).

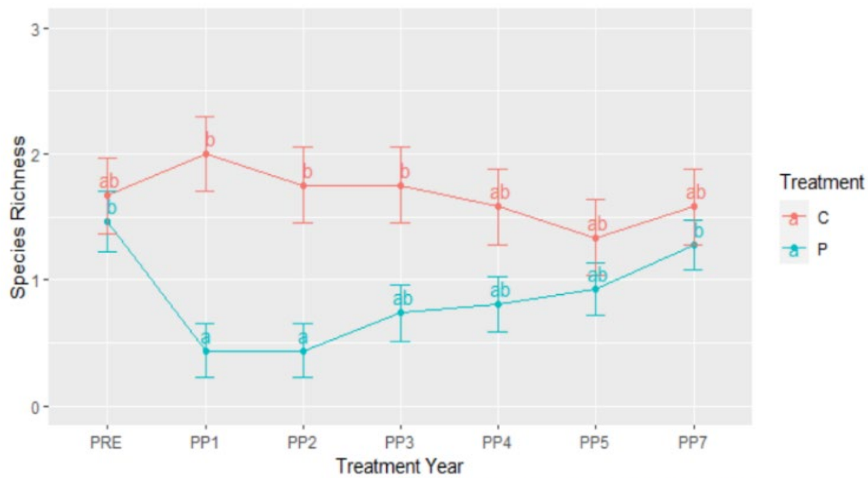


Figure 30. *Species richness in Ring Island plots over time in Control (C) and Placement Areas (P) overtime. Data presented are means ± one standard error.*

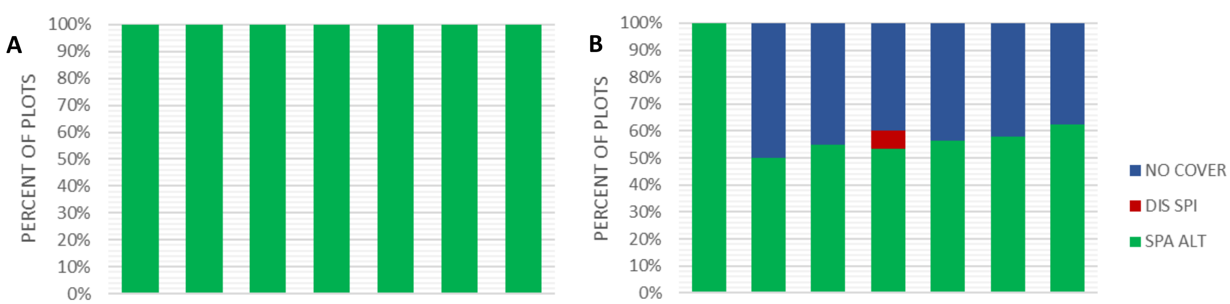


Figure 31*. Species dominance of Ring Island A) control and B) treatment plots by year. DIS SPI = Distichlis spicata; SPA ALT = Spartina alterniflora*

Avalon

There was a significant decrease in species richness at Avalon in treatment plots that started as marsh platform one-year post-placement (Figure 32; Tukey; $p < 0.001$). Both plots that started as marsh platform and plots that began as pools displayed significant recovery in species richness (Tukey; platform $p < 0.05$, pool $p < 0.001$), attaining the same level of richness as baseline conditions by PP6 (Tukey; platform $p > 0.05$, pool $p > 0.05$).

90% of control and treatment plots were dominated by short-form *Spartina alterniflora* during baseline sampling (Figure 33). This dominance remained in the control plots but changed over time in treatment plots. For treatment plots that started as marsh platform, *Salicornia* species became dominant in many plots in the first-year post-placement. In the second-year post-placement, *Distichlis spicata* became dominant in some treatment platform plots. There remained one *Distichlis spicata*- and one *Salicornia* species-dominated plot in the treatment area in PP6, with the rest of the plots containing vegetation dominated by *Spartina alterniflora*. In Avalon plots that started as pools, some *Salicornia sp.* Dominated plots appeared in PP1 and PP2, but otherwise, all plots with vegetation were dominated by *Spartina alterniflora* (Figure 33).

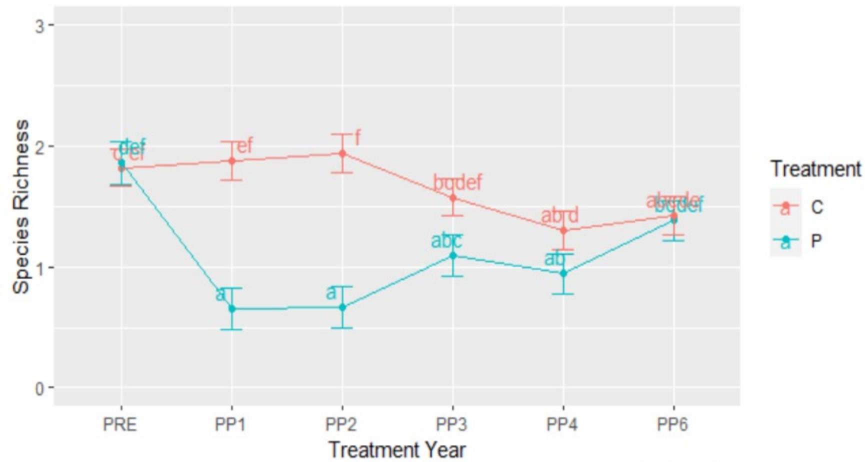


Figure 32. *Species richness in Avalon platform plots in Control (C) and Placement Areas (P) overtime. Data presented are means ± one standard error.*

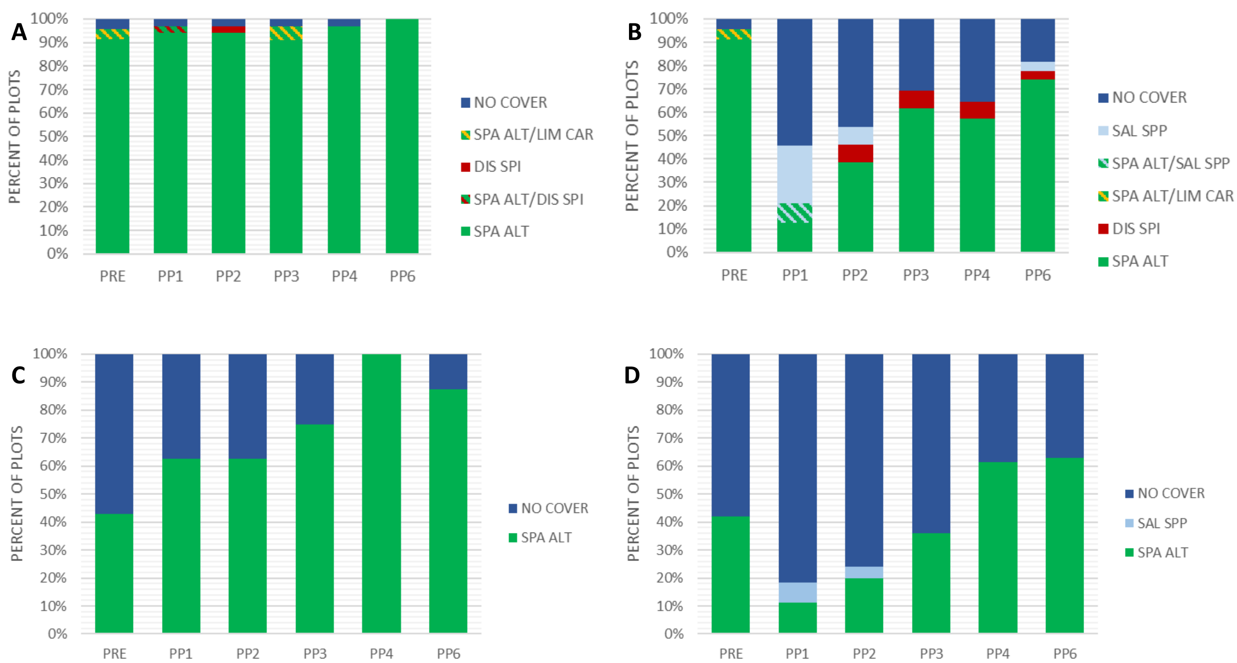


Figure 33. *Species dominance in Avalon A) platform control, B) platform treatment, C) pool control, and D) pool treatment plots by year. SPA ALT = Spartina alterniflora; LIM CAR = Limonium carolinianum; DIS SPI = Distichlis spicata; SAL SPP = Salicornia spp.*

Fortescue

A mixed effects ANOVA showed that vegetative species richness at Fortescue was significantly affected by the interaction between treatment and time ($p < 0.001$). Prior to placement, species richness was not significantly different between control and placement areas (Tukey; $p > 0.05$). Within the placement area, species richness significantly declined one-year post-placement (Figure 34; Tukey; $p < 0.001$) and then significantly increased from PP1 to PP2 (Tukey; $p < 0.05$), and from PP2 onwards species richness was no longer significantly different from baseline or control conditions. Within the control area, there has been no significant change in richness over time. Overall, these results show a significant decrease in species richness in placement areas from PRE to PP1, with recovery to baseline and control conditions by the second-year post-placement. Richness remained low in both areas with individual plots containing no more than five vegetative species.

The composition of dominant species in Fortescue plots shifted after placement in treatment plots (Figure 35). Most PRE plots were dominated by *Spartina alterniflora* with some *Spartina patens*. PP1 treatment plots were still largely dominated by *Spartina alterniflora* with one *Spartina patens*-dominated and one *Distichlis spicata*-dominated plot. In PP2, *Phragmites australis* appeared in two treatment plots. One *Salicornia* species and *Spartina alterniflora*-dominated plot appeared in the treatment area in PP4. The majority of the plots in PP6 were dominated by *Spartina alterniflora*, with *Phragmites australis, Spartina patens*, and *Distichlis spicata* plots present as well. In control plots at Fortescue, plots were roughly 60% dominated by *Spartina alterniflora* and 40% dominated by *Spartina patens*, with *Distichlis spicata* present in PRE, PP1, and PP4.

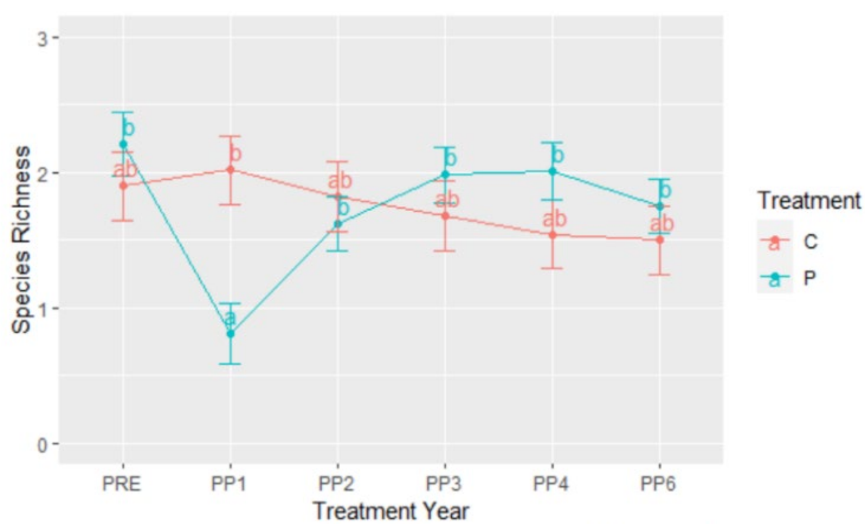


Figure 34. *Species richness in Fortescue plots in Control (C) and Placement Areas (P) overtime. Data presented are means ± one standard error.*

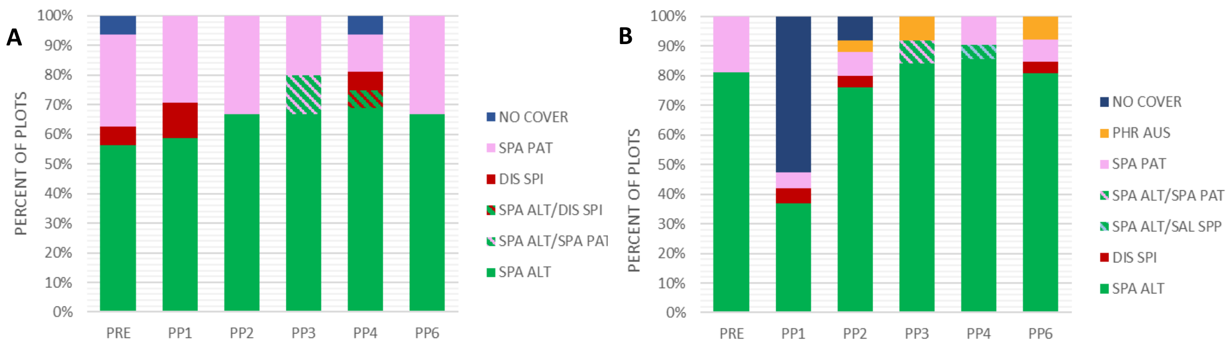


Figure 35. *Species dominance in Fortescue A) control and B) treatment plots by year. SPA ALT = Spartina alterniflora; LIM CAR = Limonium carolinianum; DIS SPI = Distichlis spicata; SAL SPP = Salicornia spp.; PHR AUS = Phragmites australis*

Multi-metric Analysis

Using program R (v. 4.2.2), robust linear regressions were performed for all possible nested models with predictor variables including placement depth (cm), elevation (ft. NAVD 88), distance to nearest tidal water body (water distance, m), distance to established vegetation (m), and penetration depth (cm). The response variables were vegetation cover (%) and stem height (cm) in permanent monitoring plots that had no vegetation after sediment placement. The sites studied were Ring Island, Avalon, and Fortescue. The Avalon dataset was separated into pool plot data and marsh platform plot data. Due to high correlation between vegetation cover and stem height, separate models were run for vegetation cover and stem height. Data was further divided by year, and models were created for PP3, PP5, and PP7 for Ring Island and PP2, PP4, and PP6 for Avalon and Fortescue (i.e., 2017, 2019, and 2021). Once the robust linear regressions were performed using the 'rlm()' function from the package 'MASS' (v. 7.3-58.1), Akaike's Information Criteria (AIC) values were calculated for each model using the 'stats' (v. 4.2.2) package to determine which model had the best fit (represented as a lower AIC value). For models that had the lowest AIC value and contained more than one variable, Variance Inflation Factors (VIFs) were calculated for each parameter to determine the extent of multicollinearity within the model using the package 'car' (v. 3.1-1). VIFs close to 1 were considered to represent non-collinearity, while values above 5 were considered to represent moderate to high collinearity. For all models with the lowest AIC value, *p*-values were determined using the package 'sfsmisc' (v. 1.1-14), and variable relationships were considered significant for value less than the alpha of 0.05.

Results

The VIFs of all linear regressions with multiple variables were found to be ≤ 3, suggesting minimal collinearity was found within selected models.

Ring Island

Vegetation Cover

In PP3, vegetation cover was not significantly related to any predictor variables measured (Table 17). However, in PP5, increased vegetation cover was significantly associated with thinner placement of sediment (slope estimate = -2.87, t = -3.04, $p < 0.05$). This relationship dissipated by PP7, as no predictor variables were found to be significantly associated with vegetation cover.

*Table 17. Robust linear regression results for Ring Island vegetation cover and stem heights in PP3 (2017), PP5 (2019), and PP7 (2021). Only significant parameters from best fit models are shown with arrows representing the direction of the regression slope estimates. Red downward arrows represent a significant negative correlation. Significance level is shown as inclusion of asterisks. Significance level: ' ' = no significance; *, p < 0.05; **, p < 0.01; ***, p < 0.001.*

Response Variable	Predictor Variable	PP3	PP5	PP7
Ring Island Vegetation Cover	*Placement Depth*		↓*	
	Elevation			
	Water Distance			
	Vegetation Distance			
	Penetration Depth			
Ring Island Stem Heights	*Placement Depth*			↓*
	Elevation	↓**	↓***	
	Water Distance	↓**	↓***	
	Vegetation Distance	↓**	↓***	
	Penetration Depth			

Stem Height

A significant increase in stem height was found in PP3 where elevation (slope estimate = -40.7, t = -7.67, $p < 0.01$), water distance (slope estimate = -0.85, t = -7.87, $p < 0.01$), and vegetation distance (slope estimate = -8.63, t = -7.22, $p < 0.01$) were smaller. Stem height remained significantly correlated with decreases in the same predictor variables in PP5 (elevation: slope estimate = -47.1, t = -7.44, $p < 0.001$; water distance: slope estimate = -0.71, t = -6.90, $p < 0.001$; vegetation distance: slope estimate = -3.93, t = -4.67, $p < 0.001$). However, only reduced placement depth was significantly correlated with stem height by PP7 (slope estimate = -0.86, t = -2.43, $p < 0.05$).

Overall Trends

Lack of correlation between vegetation cover and the predictor variables in PP2 could be due to the overall low percent cover of vegetation in the sampled plots during that time period (Figure 27). Vegetation cover at Ring Island could not be consistently predicted by any of the measured variables, suggesting that there are other unmeasured parameters that may be driving revegetation at this site. However, stem height was initially higher in areas of lower elevation and when closer to established vegetation and tidal water bodies. That difference was no longer observed in PP7, and thinner sediment placement was correlated with taller plants instead.

Avalon Pool Plots

Vegetation Cover

Within Avalon pool plots, changes in vegetation cover were not correlated with any of the measured predictor variables in PP2 (Table 18), similar to Ring Island (Table 17). Like Ring Island, lack of correlation between vegetation cover and the predictor variables in PP2 could be due to the overall low percent cover of vegetation in the sampled plots during that time period. However, by PP4, increased vegetation cover was found to be significantly correlated with decreased distance to other established vegetation (slope estimate = -2.67, t = -2.61, $p < 0.05$). By PP6, increased vegetation cover was significantly related to not only decreased vegetation distance (slope estimate = -3.52, t = -3.38, p < 0.05), but also decreased elevation (slope estimate = -29.5, t = -2.60, $p < 0.05$) and decreased penetration depth (slope estimate = -2.54, t = -2.78, $p < 0.01$).

Table 18. *Robust linear regression results for Avalon pool plot vegetation cover and stem heights in PP2 (2017), PP4 (2019), and PP6 (2021). Only significant parameters from best fit models are shown with arrows representing the direction of the regression slope estimates. Red downward arrows represent a significant negative correlation. Significance level is shown as inclusion of asterisks. Significance level: ' ' = no significance; *, p < 0.05; **, p < 0.01; ***, p < 0.001.*

Response Variable	Predictor Variable	PP2	PP4	PP6
Avalon Pool Plot Vegetation Cover	*Placement Depth*			
	Elevation			↓*
	Water Distance			
	Vegetation Distance		↓*	↓*
	Penetration Depth			↓**
Avalon Pool Plot Stem Heights	*Placement Depth*			
	Elevation			
	Water Distance			
	Vegetation Distance		↓*	↓***
	Penetration Depth			

Stem Heights
Similar to vegetation cover, PP2 stem heights were not associated with any predictor variables, but measurements from PP4 and PP6 demonstrate that plants were taller when in closer proximity to established vegetation communities (PP4: slope estimate = -2.66, t = -3.96, $p < 0.001$; PP6: slope estimate = -2.61, t = -4.22, $p < 0.001$).

Overall Trends
These results demonstrate that vegetation re-establishes and is taller in areas that are in closer proximity to existing vegetation in areas that started as unvegetated pools. Interestingly, reduced penetration depth was associated with an increase in vegetation cover in PP6, which suggests that firmer TLP sediments in former pool plots may have improved restoration results. (Figure 28C).

Avalon Platform Plots

Vegetation Cover
Increases in vegetation cover in PP2 were significantly associated with decreasing distance from nearby waterbodies (slope estimate = -0.28, t = -2.50, $p < 0.05$; Table 19). However, neither PP4 nor PP6 vegetation cover values were significantly related to any of the measured predictor variables.

Stem Heights
Similar to vegetation cover trends in PP2, stem heights of vegetation in platform plots increased with decreasing distance to the nearest waterbody (slope estimate = -0.36, t = -2.65, $p < 0.05$). However, stem heights were additionally found to increase with increased elevation (slope estimate = 21.3, t = 2.10, $p < 0.05$). In PP4, decreasing distance to the nearest established vegetation community was found to be the main predictor of stem height (slope estimate = -3.65, t = -2.61, $p < 0.05$). By PP6, both elevation (slope

estimate = 33.3, t = 3.15, $p < 0.01$) and penetration depth (slope estimate = 8.77, t = 3.40, $p < 0.05$) were positively associated with stem height. Decreasing distance to the nearest established vegetation community was additionally found to be significantly correlated with increased stem height (slope estimate = -5.30, t = -2.69, $p < 0.05$).

Overall Trends

Vegetation cover could not be consistently predicted by any measured variables, suggesting that different forces are acting on the marsh platform of Avalon compared to the pools. Stem heights were overall associated with a decrease in elevation and an increase proximity to established vegetation communities. in PP6, penetration depth was also associated with taller vegetation.

Table 19. *Robust linear regression results for Avalon platform plot vegetation cover and stem heights in PP2 (2017), PP4 (2019), and PP6 (2021). Only significant parameters from best fit models are shown with arrows representing the direction of the regression slope estimates. Red downward arrows represent a significant negative correlation, while green upward arrows represent a significant positive correlation. Significance level is shown as inclusion of asterisks. Significance level: ' ' = no significance; *, p < 0.05; **, p < 0.01; ***, p < 0.001.*

Response Variable	Predictor Variable	PP2	PP4	PP6
Avalon Platform Plot Vegetation Cover	*Placement Depth*			
	Elevation			
	Water Distance	↓*		
	Vegetation Distance			
	Penetration Depth			
Avalon Platform Plot Stem Heights	*Placement Depth*			
	Elevation	↑*		↑**
	Water Distance	↓*		
	Vegetation Distance		↓*	↓*
	Penetration Depth			↑*

Fortescue

Vegetation Cover

In PP2, no predictor variables were found to be associated with vegetation cover changes (Table 20). In PP4, both water distance (slope estimate = 0.50, t = 4.41, $p < 0.001$) and vegetation distance (slope estimate = -50.1, t = -2.72, $p < 0.01$) were shown to be positively and negatively correlated with vegetation cover, respectively. These relationships were not found in PP6.

Stem Height

Stem height was found to be significantly higher with greater penetration depth in PP2. However, in both PP4 and PP6, only elevation was found to be negatively correlated with stem heights (PP4: slope estimate = -58.5, t = -3.73, $p < 0.01$; PP6: slope estimate = -29.0, t = -4.26, $p < 0.001$).

Overall Trends

Vegetation cover could not be consistently predicted by any measured variables, suggesting that different forces are acting on the marsh platform at Fortescue. Much of the revegetation occurred between PP2 and PP6, suggesting that the significant relationship between vegetation cover and proximity to water and existing vegetation was important at Fortescue. By PP6 all plots had at least some vegetation, making proximity to existing vegetation 0 m for all plots. Only stem heights were found to be more consistently correlated with elevation in later years of monitoring. Similar to Ring Island (Table 17), but unlike Avalon (Table 18 & 19), lower elevation resulted in taller stems.

Table 20. *Robust linear regression results for Fortescue vegetation cover and stem heights in PP2 (2017), PP4 (2019), and PP6 (2021). Only significant parameters from best fit models are shown with arrows representing the direction of the regression slope estimates. Red downward arrows represent a significant negative correlation, while green upward arrows represent a significant positive correlation. Significance level is shown as inclusion of asterisks. Significance level: ‘ ’ = no significance; *, p < 0.05; **, p < 0.01; ***, p < 0.001.*

Response Variable	Predictor Variable	PP2	PP4	PP6
Fortescue Vegetation Cover	*Placement Depth*			
	Elevation			
	Water Distance		↑***	
	Vegetation Distance		↓**	
	Penetration Depth			
Fortescue Stem Heights	*Placement Depth*			
	Elevation		↓**	↓***
	Water Distance			
	Vegetation Distance			
	Penetration Depth	↑*		

Trends of All Sites

Overall, higher vegetative cover was most often associated with closer proximity to existing vegetation. This tracks with the observation made at all three sites that vegetation spread from existing vegetation, either from the edges of placement or from the few clumps of plantings that thrived within the sites.

Stem heights were most closely associated with lower elevation and closer proximity to tidal water bodies and existing vegetation. Tall-form *Spartina alterniflora* plants is known to grow in lower elevations and closer proximity to creeks (Howes et al. 1986, Bertness and Ellison 1987; Tyler and Zieman 1999). It is less clear why there would be a strong relationship between plant height and proximity to other plants except that vegetation is likely to inhabit areas with fewer environmental stressors and those stressors can also stunt *S. alterniflora* plants.

These findings suggest that having smaller placement areas or placement areas with a higher edge to interior ratio may decrease vegetation recovery time.

Conclusions

- A look at habitat proportions (vegetated, pool, and bare) over time was useful for a general understanding of how the sites changed. All sites had a large initial conversion of vegetated plots to unvegetated plots and then saw a steady shift back to high proportions of vegetated plots. By 2021: 1) The proportion of vegetated plots at Ring Island was lower than in control sites and in baseline conditions, but there is an increasing trend over time. 2) At Avalon, there was an increase in vegetated plots from the baseline, in part because some pool plots became vegetated, but remained lower than in control sites, but there is an increasing trend over time. 3) Fortescue had an initial decrease of 80% of its vegetated plots but was back up to 100% by 2021.

- Percent cover is a density estimate that gives a closer look at how well plots are revegetating. All sites had very low percent cover in the first growing season post placement (3-11%). Since then, the sites have had highly variable recovery and none of the sites show an uplift from baseline conditions. There does not appear to be a set of environmental variables (at least among those evaluated) that consistently correlated with the percent vegetation cover recovery at all three sites. Therefore, site-specific factors appear to be more important than overall wetlands ecosystem factors.
- Despite the increasing trend in the proportion of plots that are vegetated at Ring Island, the trend in vegetation cover showed that the density of plants in placement areas stagnated in the seven years since placement and has not shown an uplift compared to baseline or control site conditions. At Avalon, after six years of recovery, vegetation cover in placement plots increased and was no longer different from vegetation cover in control plots or baseline conditions. At Fortescue, placement plot cover increased in the years after placement and is no longer different from control sites, but vegetation cover remained significantly lower than baseline conditions.
- Percent cover in control sites at all three projects declined over time.
- Correlations between percent cover and other environmental variables can help explain why the sites had different plant recovery rates and suggest improvements for planning future projects.
- The differences in recovery between the three sites can be attributed to a variety of potential factors including differences in how the projects were constructed and the environmental characteristics of the sites. We found three main driving forces affecting revegetation rates: 1) Short-term vegetation recovery was higher in sites that had less sediment placement and lower elevations; 2) vegetation was denser in areas near existing vegetation, suggesting that higher edge-to-placement ratios may speed up the recovery rate; and 3) while final elevation is important in long-term plant recovery (as evidenced at Ring Island), placement depth was not (as evidenced by vegetation growth in Avalon's pool plots).
- All three project sites experienced an initial decline in plant species richness. The decline was minimal due to the inherent low overall species richness (1-3 species) at the sites. It is encouraging that no native species were completely lost from the project sites, other than any subaquatic vegetation at Avalon due to the filling of pools.
- Treatment plots primarily revegetated with *Spartina alterniflora*, a low marsh species, after sediment placement despite our initial goal of facilitating high marsh species habitat (characterized by *Spartina patens*).
- This study documents a slight shift in species composition over time at the three sites after sediment addition, with placement plots being initially dominated by early colonizers like *Distichlis spicata* and *Salicornia* sp., and subsequently dominated by *Spartina alterniflora*.
- The invasive *Phragmites australis* was only found in plots located near the footprint of the constructed Fortescue dune where the elevation and hydrology are supportive of this species. *Phragmites australis* colonization of placement areas has been limited to only two treatment plots; however, as it is present within the Fortescue Dune and Ring Island ENH, it is being closely monitored. When *P. australis* is detected, it is treated with herbicide according to an adaptive management plan.
- These sites should continue to be monitored to determine the amount of time needed for all three sites to be enhanced beyond their baseline and control conditions and to further understand the

major factors influencing vegetation recovery.

- Based on these results, we can answer the following monitoring questions: 1) Does the marsh ecosystem recover or show uplift within two to three years of dredged material placement? (And the related, "How long does it take for the marsh to recover and achieve uplift?"); 2) Did differences in structural factors correlate with recovery and uplift?; and 3) What parameters should be included in a marsh enhancement monitoring program, and how long should the post-placement program be conducted?
    - 1) Significant uplift of the ecosystems was not shown within three years post placement at any of the sites. Ring Island vegetation cover was decreased significantly by dredged sediment placement. The percentage of cover did not significantly increase within two to three years of placement. Species richness was found to be insignificantly different from PRE conditions, although it remained low overall after three years. Avalon vegetation cover was found to be insignificantly different after three years compared to PRE conditions; however, platform plots were still much lower in coverage than PRE conditions. Pool plots were found to be similar, but no uplift was found after a few years post-placement. Species richness remained significantly lower than PRE conditions by PP3. Although the percent cover of vegetation did increase within three years after placement at Fortescue, the percent cover remained insignificantly different from PRE conditions. Species richness was found to be similar by the third year post-placement compared to PRE conditions, but it did not exceed previous conditions. Based on all years of vegetation monitoring, we can conclude that significant uplift has not occurred within six to seven years post placement; however, several parameters are on an upward trend. Significant uplift may take a decade or longer to be seen.
    - 2) At Ring Island, no variables were significantly correlated with vegetation cover due to a lack of vegetative recovery for the first few years. However, placement depths greater than 12 cm deep resulted in < 20% vegetation cover. In PP5, increased vegetation cover was associated with thinner placement, lower elevations, and closer proximity to existing vegetation. In PP7, increased cover remained associated with thinner placement and lower elevations. At Avalon, from 2016 through 2019, no variables were significantly correlated with vegetation cover in plots that started as marsh platform. By 2021 (PP6), plots that started as platforms compared to plots that started as pools had greater cover the closer they were to existing vegetation and tidal water bodies. Vegetation cover also increased with greater sediment placement depths and where the soil was firmer. At Fortescue, in PP2, thicker sediment was significantly correlated with lower vegetation cover. That trend disappeared in PP3, suggesting that vegetation can recover in areas of deeper sediment placement.
    - 3) Based on our findings, we suggest increasing the frequency of monitoring and expanding the monitoring area beyond the placement and control areas. We were able to determine valuable metrics as predictors of TLP success, with differences between vegetative cover and stem height response. Increase in vegetation cover was most closely associated with closer proximity to nearby established vegetation, while stem height increases were most closely related to lower elevation, closer proximity to tidal water

bodies, and reduced distance to nearby established vegetation. Other predictors, such as placement depth and penetration depth were not found to be consistently related to either vegetation cover or stem height. Although we were able to determine valuable metrics, we have found that there are further environmental effects that were not captured within our monitoring schema. Monitoring outside the typical study area may provide context for general environmental effects, such as water table elevation shifts, nutrient concentrations in runoff, and microbial community shifts. Additionally, post-placement monitoring timing should strive towards at least a decade due to the slow pace of recovery after sediment placement.

6. GIS Habitat Characterization Analysis

William Smith[1], Brittany Wilburn[1,2], Alex Ferencz[3], Steven Jacobus[1], Metthea Yepsen[1]
[1]New Jersey Department of Environmental Protection, [2]Drexel University, [3]Stockton University

As noted in Section 5, vegetation is an important component in the maintenance of long-term salt marsh health. In addition to the permanent vegetation plots, aerial imagery was used to evaluate overall shifts in habitat cover to monitor the sites on a landscape level. Although vegetation plots offer a more detailed evaluation of plant species and vigor, they do not capture the large-scale changes resulting from TLP. Additionally, habitat cover allows for the evaluation of hydrologic and topographic effects on vegetation cover. Classification of aerial imagery of the projects can demonstrate the direct effects of landscape changes on the ground cover type. Object-based image analysis (OBIA) is a method of classification that uses machine learning to group similar pixels together and treat these groups as objects rather than evaluating the image on a pixel-by-pixel basis. Important metrics, such as the rate of vegetation recolonization and percent cover of vegetation, or other classification types can be evaluated using OBIA of aerial imagery collected at key time points during the project.

Monitoring Design

Object-Based Image Analysis

An object-based mapping workflow was utilized to generate the landscape data used in this project. Object-based mapping offers a powerful and efficient means of classifying imagery by grouping the pixels of imagery into objects rather than analyzing the image on a pixel-by-pixel basis (Addink and Coillie, 2010). Segmentation and training sample data are two key components of the workflow for this analysis. Segmentation groups the pixels into simplified objects based on spectral detail, shape, and size of the object neighborhood (Addink and Coillie, 2010). Alternatively, pixel-based classifiers classify pixels based on spectral detail and texture (Congalton et al., 2017). The object-based method was preferred over the pixel-based method because of the high-resolution imagery and general classes used to create the landscape datasets. The segmented image was used as the basis for training sample data to assign labels (or classes) to these objects. This served as a type of supervised classification scheme (Congalton et al., 2017). The labels used for the classification schema included “Non-Veg,” “Vegetation,” and “Water.” The resulting datasets were put through a machine learning algorithm in ESRI’s ArcGIS Pro v2.6 and 2.7.3 (ESRI, 2020 and 2021) to produce a classified image of the marsh.

For this analysis, nine different sets of imagery were processed and assessed. Aerial imagery was obtained for each control and impact site in Avalon, Ring Island, and Fortescue. NJDEP drone imagery (captured using a DJI Phantom 4 PRO quadcopter with a 1-inch 20-megapixel RGB camera) was used in combination with aerial imagery from NAIP and additional sUAS imagery from The Nature Conservancy (Table 21). A description of NAIP imagery collection methods can be found at USDA NAIP Imagery Program. Imagery collected in 2013, 2016, and 2020 represent the baseline, “as built”, and current conditions of each site, respectively. Imagery included only red-, green-, and blue-wavelengths (RGB), so analyses focused on detecting the presence and coverage of vegetation rather than health indices or other multispectral imagery. Additionally, the 2016 datasets do not include Control areas.

A segmented image and training sample data were derived for each of the 2013, 2016, and 2020 imagery datasets because of the differing resolution, spectral signatures, and collection conditions between the three datasets. The segmented image and training samples were then used with the Support Vector Machine (SVM) machine learning algorithm to classify the presence of each class on the marsh. Total acreage per testing site as well as percent coverage of each class were calculated based on the grouping of "Non-Veg" categories (consisting of "Water" & "Non-Veg" classes) and "Vegetation", respectively.

Table 21. *Summary of aerial imagery sources and parameters.*

Time Period	Year of Collection	Resolution	Source	Collection Method
Baseline	2013	39-inch	NAIP	Aerial Collection
"As Built"	2016	6-inch	The Nature Conservancy	sUAS*
Current	2020	3-inch	NJDEP	sUAS
*sUAS = Small Unmanned Aircraft System				

Segmentation and Training Datasets

The creation of the segmented image reduced the variability and spectral detail the original orthomosaic provided. This streamlined the classification of vegetation, soils, and water because the machine learning algorithm used training samples to predict the classification based on the similarity of spectral and spatial detail of adjacent object neighborhoods instead of using the complex spectral detail and texture of pixels (Congalton et al., 2017). The main goal of creating a segmented image is to create objects that are spectral and spatially distinct from other objects while preserving variability within each object (Congalton et al., 2017). Using ArcGIS Pro's "Segmentation" tool, a low spectral detail and a low spatial detail were favored so that the different species of vegetation were merged into similar object neighborhoods, but not so low that it merges with the features of the other classes (ESRI, "Segmentation"). The simplified spectral information reduced the complexity of the model and the number of training samples needed because of this similarity in object neighborhoods within each class. The training sample data were composed of polygons manually drawn on the map to represent each class of the marsh. The amount of each training class polygon depended upon the size of each testing site, the presence of each class on the map, spectral detail of the imagery, and conditions when each imagery dataset was collected. Because of this, the amount of training data for each class differs from year to year and from site to site. Each training sample must also represent each class consistently, exclusively, and completely to avoid adding confusion into the model (Congalton et al., 2017) Training the data to get a representative sample of the spectral variability in each class was preferred over taking equal samples to ensure the classes were correctly classified without overfitting the model.

Classification and Review

The segmented image and training samples then were used to classify the imagery using the Support Vector Machine (SVM) machine learning algorithm using ArcGIS Pro's "Classify" tool. Once the image was classified, a manual review of the data was done within each testing site boundary to ensure thematic accuracy. Areas outside of the site boundaries were clipped out and did not receive edits. Classification errors were corrected using the Pixel Editor tool in ArcGIS Pro. The minimum mapping unit used for each feature was approximately 1 sq. ft. Because of the differing resolutions of the imagery, the classified raster output was edited at different scales. Data derived from the 3-inch 2020 imagery datasets were edited between a 1:50 and 1:300 scale. Data derived from the 6-inch 2016 datasets were edited between 1:100 and 1:300. Data derived from the 1-meter 2013 imagery datasets were edited between 1:800 and 1:1,100. This was to ensure the accuracy of feature boundaries between classes as well as to correct errors within each feature boundary.

Other features not represented in the class labels were also present in the imagery. This included debris, wrack, field and flight crew members, field equipment, other ongoing projects present on the marsh, fencing, and a boat. These features were often initially classified as “Non-Veg,” but were reclassified as the class of the feature beneath it based on site knowledge. In areas of dense wrack, it was chosen to classify the area as “Non-Veg” due to a combination of unknown underlying cover and the possibility that the wrack could cause vegetation die-off underneath its dense layers (Bertness and Ellison, 1987). However, if the wrack wash was sparsely distributed across the marsh, it was reclassified as the underlying class. Once the datasets were corrected, they were converted from raster format to vector form, clipped to the testing site boundary, and enriched with the attributes of other project data. The analysis and display of the final classification, acreage of each class, and percent coverage were based on these added attributes.

Calculations

Based on the image classification, the acreage and percent coverage of each testing site were calculated using each testing site buffer and site type (control group or experimental group), respectively. One caveat is this image classification was based on the visual presence of each class and does not take recent tide and rain events into account when digitizing each class. Depending on when the imagery datasets were collected, different water levels may be present on the marsh. Therefore, although the water and bare soil features were classified as their respective classes, they were grouped using the “Non-Veg” subcategory in the final calculations because of these fluctuations in tidal and pooled water. The final comparison was between the “Non-Veg” class, which was composed of the “Water” and “Non-Veg” main classes, and the “Vegetation” class. These calculations will be integrated into the broader analysis of this project to assess the success of using dredge material as a means of helping salt marsh habitat.

Accuracy Assessment

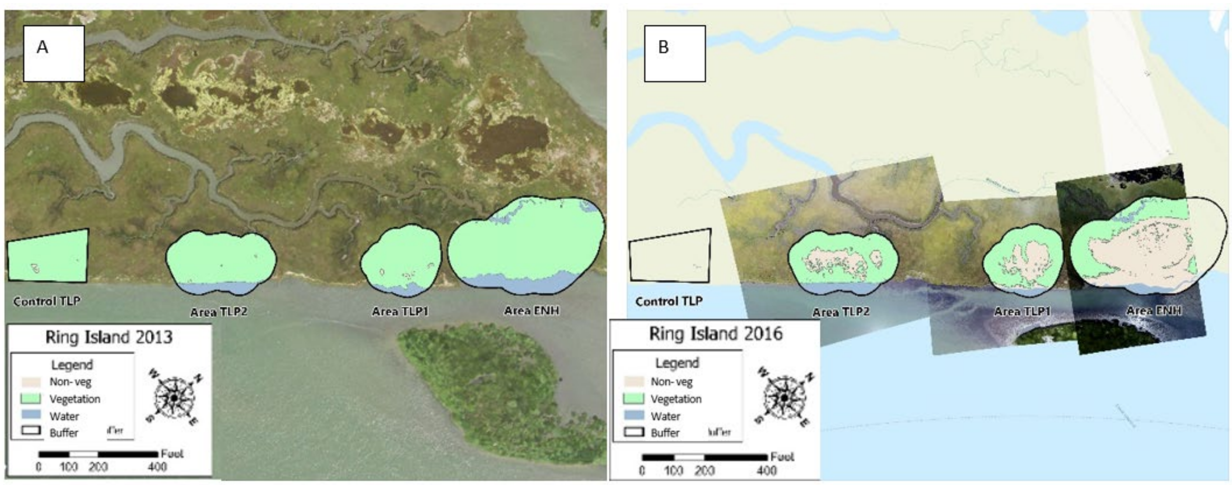
The thematic accuracy assessment was conducted with suggestions outlined in *Assessing the Accuracy of Remotely Sensed Data* (Congalton and Green, 2019). Overall accuracy was the primary accuracy statistic used to gauge the thematic accuracy of each dataset. As a general accuracy standard, 90% accuracy was designated as the minimum allowable accuracy for each collection. Anything below this was considered unacceptable and further editing of the classified data was done. Several checkpoints were designated for each testing site due to the differing size of the three testing sites. Due to the disproportionate presence of each class on each of the three testing sites, the checkpoints were divided up proportionately based on the overall percentage of area each class possesses (Congalton and Green, 2019). For instance, if a class takes up 30% of the overall area of the testing site, 30% of the checkpoints used to test the thematic accuracy of that testing site will be used for that class.

Results

Ring Island

Classification of Ring Island in 2013 revealed baseline conditions of thorough vegetation, with a total percent cover of 87.5%, or 8.97 of the 10.25 acres (ac) (Figure 36A and 37). Within the control area, 99.1% was vegetated (1.47 ac), while pre-placement areas had 85.6% plant cover (7.52 ac) (Figure 38). Classification of 2016 imagery, collected two years post-placement, demonstrated significant decreases in vegetation cover, with patterns of centralized non-vegetation patches and rings of vegetation on the outer edges of the treatment areas (Figure 36B).

However, six years post-placement in 2020, there are clear signs of revegetation within both the TLP areas and the ENH area. Vegetation began to fill in the center of the ENH treatment area, and non-vegetated patches of the TLP areas were reduced (Figure 36C). 2020 levels of vegetation do not match baseline conditions, with only 69.1% total vegetation cover (7.12 ac, a loss of 3.13 ac; Figure 37). However, within the control area, there is also an increase in the non-veg cover, rising to 7.8% (or 0.12 ac) from its baseline condition of 0.9% (or 0.01 ac), resulting in a less contiguous and patchier habitat (Figure 38). This suggests



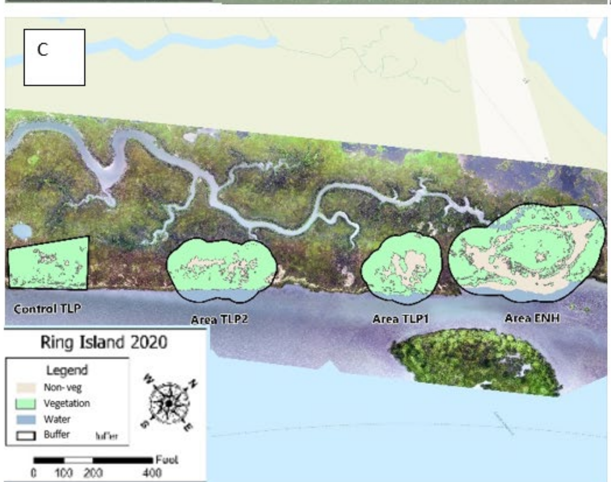


Figure 36. *Classification of vegetation and water cover of Ring Island A) pre-placement, B) 2 years post-placement, and C) six years post-placement of dredged material.*

that there are ongoing conditions causing vegetation die-off in the overall site. Broadly, Ring Island placement areas revegetated at a rate of 0.43 ac per year (3.7% of the site per year) based on vegetation change from 2016 to 2020. This slow rate of revegetation could be a result of higher elevations (see Section 4: Water Level Monitoring and Section 5: Vegetation from Plot Based Monitoring).

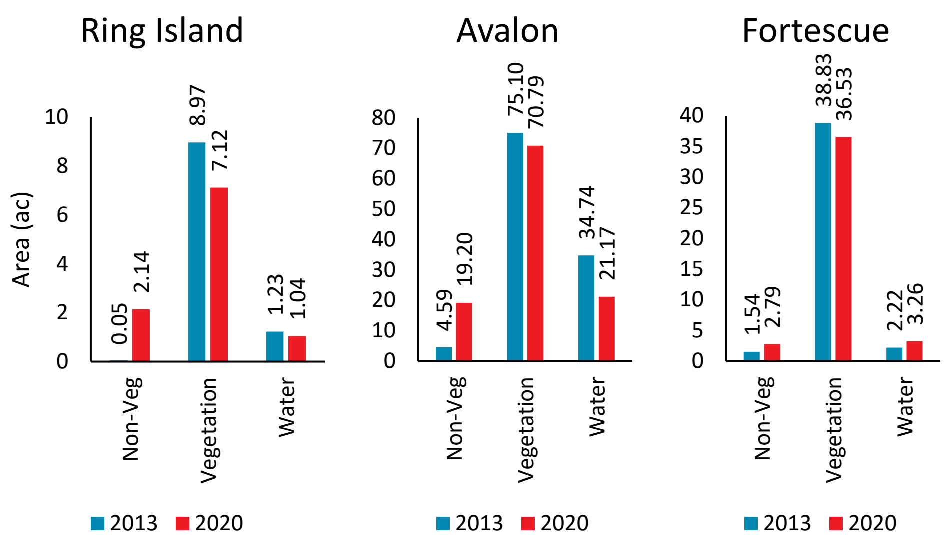


Figure 37. *Change in total (i.e., control and placement areas combined) non-vegetative, vegetative, and water cover square acreage in Ring Island, Avalon, and Fortescue from pre-placement to multiple years post-placement of dredged material.*

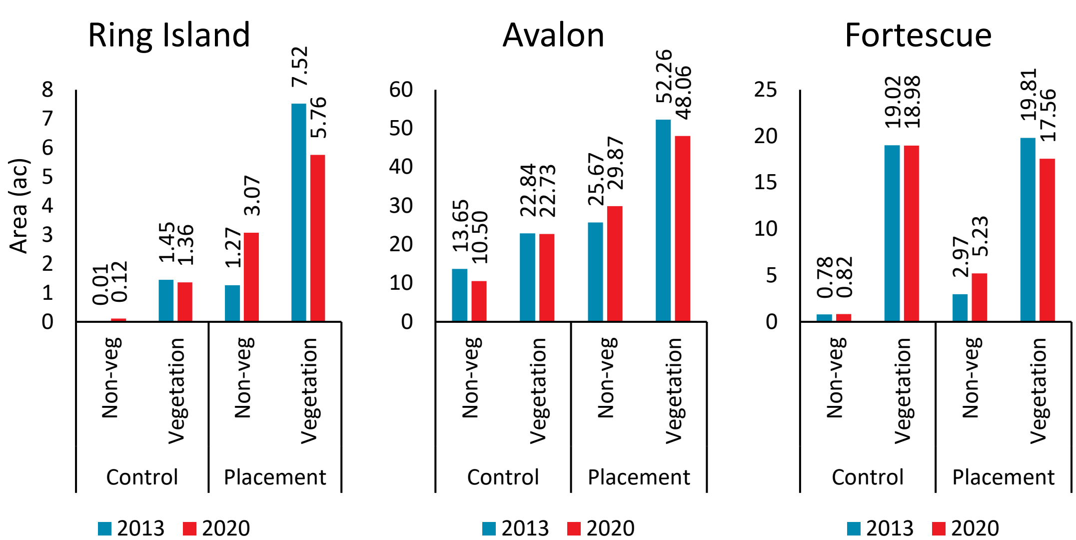


Figure 38. *Change in vegetative cover square acreage in Ring Island, Avalon, and Fortescue control and placement areas from pre-placement to multiple years post-placement of dredged material.*

Avalon

Baseline classification of Avalon control and pre-placement areas show that the marsh was heavily influenced by ponding, with open water habitat constituting 30.4% (or 34.74 ac) of the total area and vegetation covering 65.6% (75.1 ac; Figures 39A and 37). Control and pre-placement areas had similar ratios of vegetated to non-vegetated area, with 62.6% (22.84 ac) and 67.1% (52.26 ac) vegetation coverage, respectively (Figure 38).

Classification of the 2016 Avalon habitat (collected the summer after Phase 2 of placement) shows significant retention of ponds within the placement areas (Figure 39B). Similar to the Ring Island site, vegetation remained as a ring outside of the placement areas, with non-vegetated areas and ponding centered within the treatment areas.

Four years post-placement in 2020, classification shows substantial revegetation of the placement areas. The final percent cover of vegetation in placement areas reached 61.7% (48.06 ac), nearly reaching baseline conditions measured in the 2013 dataset (Figures 39C and 37). Ponding and non-vegetated areas were more mosaiced when compared to the larger contiguous ponds found in the 2013 classification. Control sites remain steady in terms of vegetation cover at 22.7 ac in 2020 compared to the 22.8 ac in

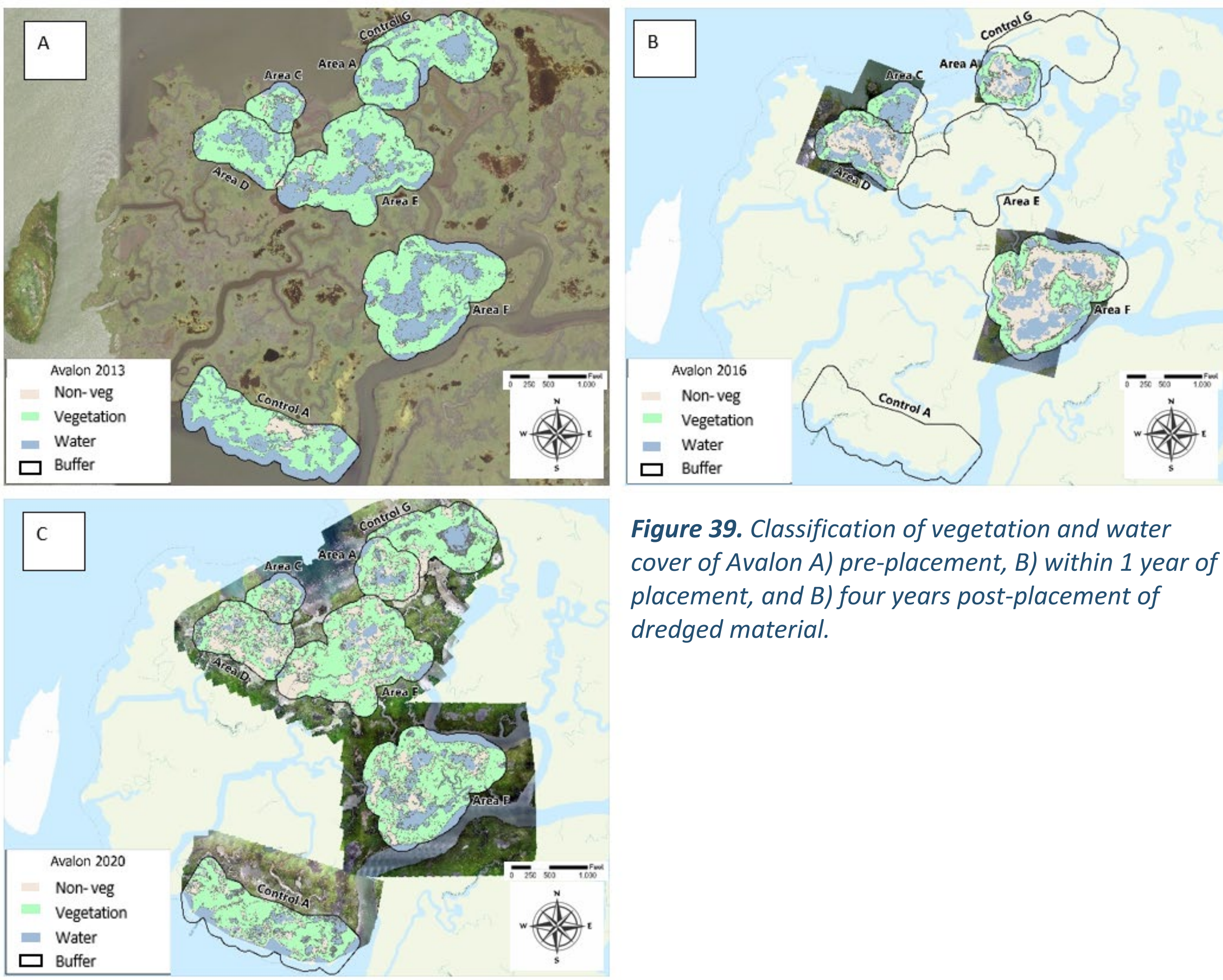


__Figure 39.__ Classification of vegetation and water cover of Avalon A) pre-placement, B) within 1 year of placement, and B) four years post-placement of dredged material.

2013 (68.4% vs 62.6% cover respectively; note, that percentages are different due to clipping of the Control Area A by lack of full imagery coverage of 2020 imagery; Figure 38). Within Avalon placement areas, we found that the sites revegetated at a rate of 8.01 ac per year (7.0% of the site per year) from 2016 to 2020. This rate of revegetation was much faster than Ring Island and could be attributed to the overall lower final elevation (Tables 5 and 8 in Section 3: Elevation).

Fortescue

2013 classification of the habitat within the Fortescue site shows 91.2% (38.83 ac) of the total site was vegetated, with minimal ponding and other non-vegetated cover (water: 3.6%, 1.54 ac; non-veg: 5.2%, 2.22 ac; Figure 37 and 40A). Control areas had a slightly higher percent cover of vegetation of 96% (19.02 ac) compared to pre-placement areas with a percent cover of 87% (19.81 ac; Figure 38).

2016 classification of Fortescue included placement areas just after treatment with dredged material, but before the dune and beach restoration. Placement areas showed significantly higher proportions of non-vegetated areas as expected; however, there were additional developments of ponding within placement areas that were not present in the 2013 classification (Figure 40B).

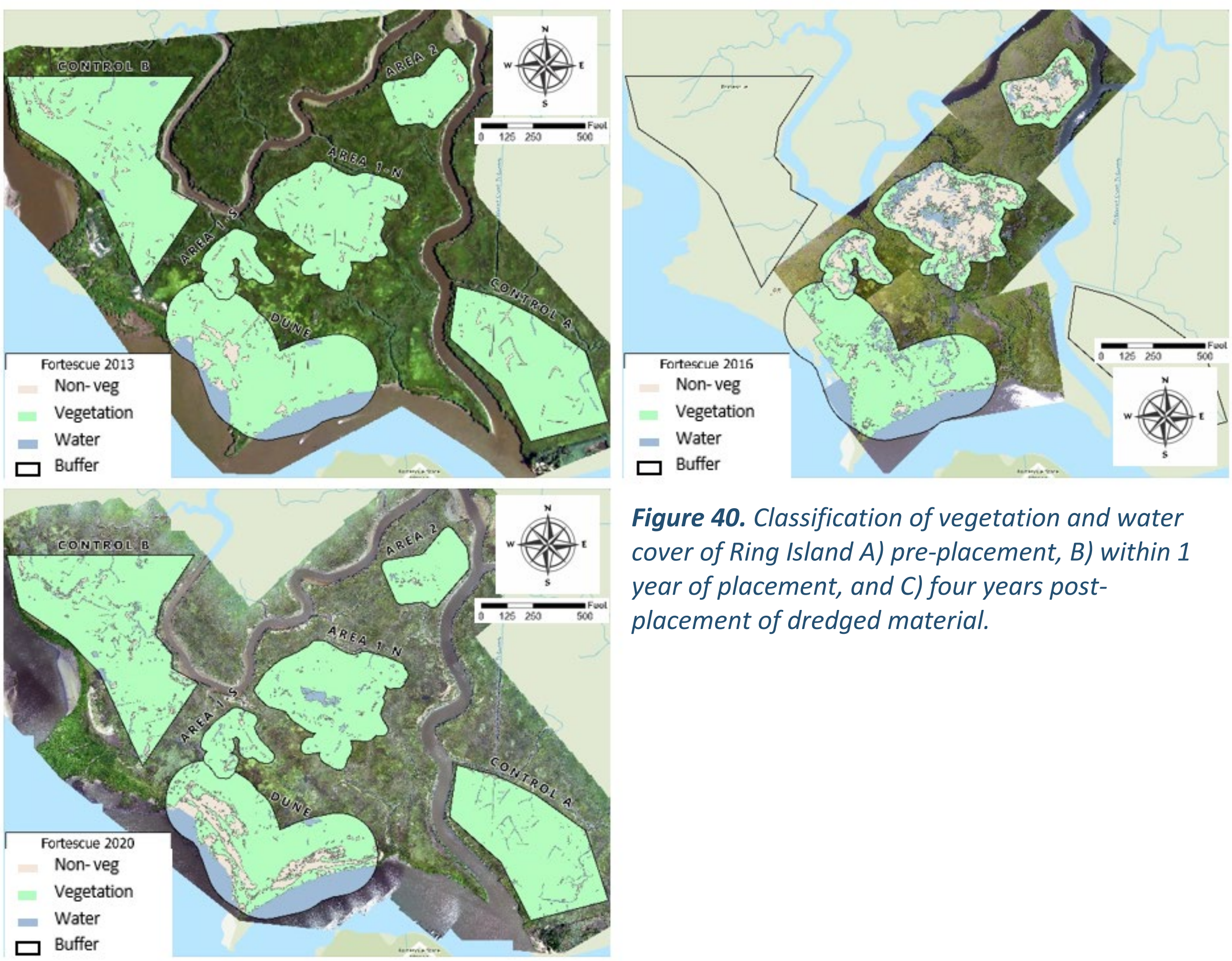


Figure 40. *Classification of vegetation and water cover of Ring Island A) pre-placement, B) within 1 year of placement, and C) four years post-placement of dredged material.*

By 2020 (four years post-placement and three-years post-restoration of the dune and beach areas), much of the area that was unvegetated in 2016 had grown in (Figure 40C). Much of the dune enhancement site was still unvegetated, but some of the plantings had begun to increase in cover. Overall, the final vegetation cover did not meet baseline conditions, reaching 85.8% (36.53 ac) compared to baseline levels of 91.2% (38.83 ac; Figure 37). Additionally, some of the newly developed ponding remained, reducing the vegetative cover slightly. This increase in ponding, in conjunction with higher elevations similar to Ring Island (Tables 5 and 9 in Section 3: Elevation), may have contributed to a reduced rate of revegetation (in comparison to Avalon) of 1.16 ac per year (4.3% of the site per year) from 2016 to 2020. Additionally, the analysis included the dune restoration, where much of the vegetated cover converted to unvegetated cover from 2016 to 2020, thus potentially skewing the vegetation rate lower. Plot-based monitoring (see Figures 27, 28, and 29 in Section 5: Vegetation from Plot Based Monitoring), however, found that Fortescue revegetated much faster within the placement areas than Avalon and Ring Island. These differences in revegetation rates between this classification and the findings of the plot-based monitoring could be largely attributed to the inclusion of the dune restoration, which was much slower to revegetate over time. Permanent plots were not placed in the dune, and vegetation was not examined in detail in this area.

Accuracy Assessment

Due to the varying sizes of each testing site, a different number of checkpoints were used to evaluate the accuracy of the classification but were consistent from year to year (Table 22). Across all sites, we were able to achieve >90% classification accuracy.

In addition to manually checking the classification, percent cover and habitat proportions defined by the classification workflow were compared to what was found in the permanent vegetation plots. Figure 41 reflects the percent cover vegetation, or vegetation density, as calculated by permanent vegetation plots and drone imagery (see Section 5: Vegetation from Plot Based Monitoring for more detailed results). Classification from drone imagery was found to have higher estimates of the percent cover of vegetation compared to the on-the-ground permanent plots. In part, this may be attributed to the different scales of analysis, as the drone imagery was evaluating land cover over multiple acres while the vegetation plots could be analyzed in detail by the field team. It is also likely that the plots-based cover estimates underestimate cover due to their locations away from the edge of placement areas where vegetation recovered first.

Habitat proportion was much more easily comparable between the vegetation plots and drone imagery (Figure 42). This landscape-scale metric was found to be very similar between the two methods, suggesting that drone imagery is a good proxy for measuring habitat cover ("Non-Veg," "Vegetation," and "Water").

Table 22. *Thematic accuracy assessment of habitat classification at each site.*

Site Name	Year	Number of Checkpoints	Overall Accuracy
Ring Island	2013	125	95.5%
	2016	125	92%
	2020	125	98.5%
Avalon	2013	200	96%
	2016	200	96.5%
	2020	200	99%
Fortescue	2013	200	97.5%
	2016	200	96%
	2020	200	99.5%

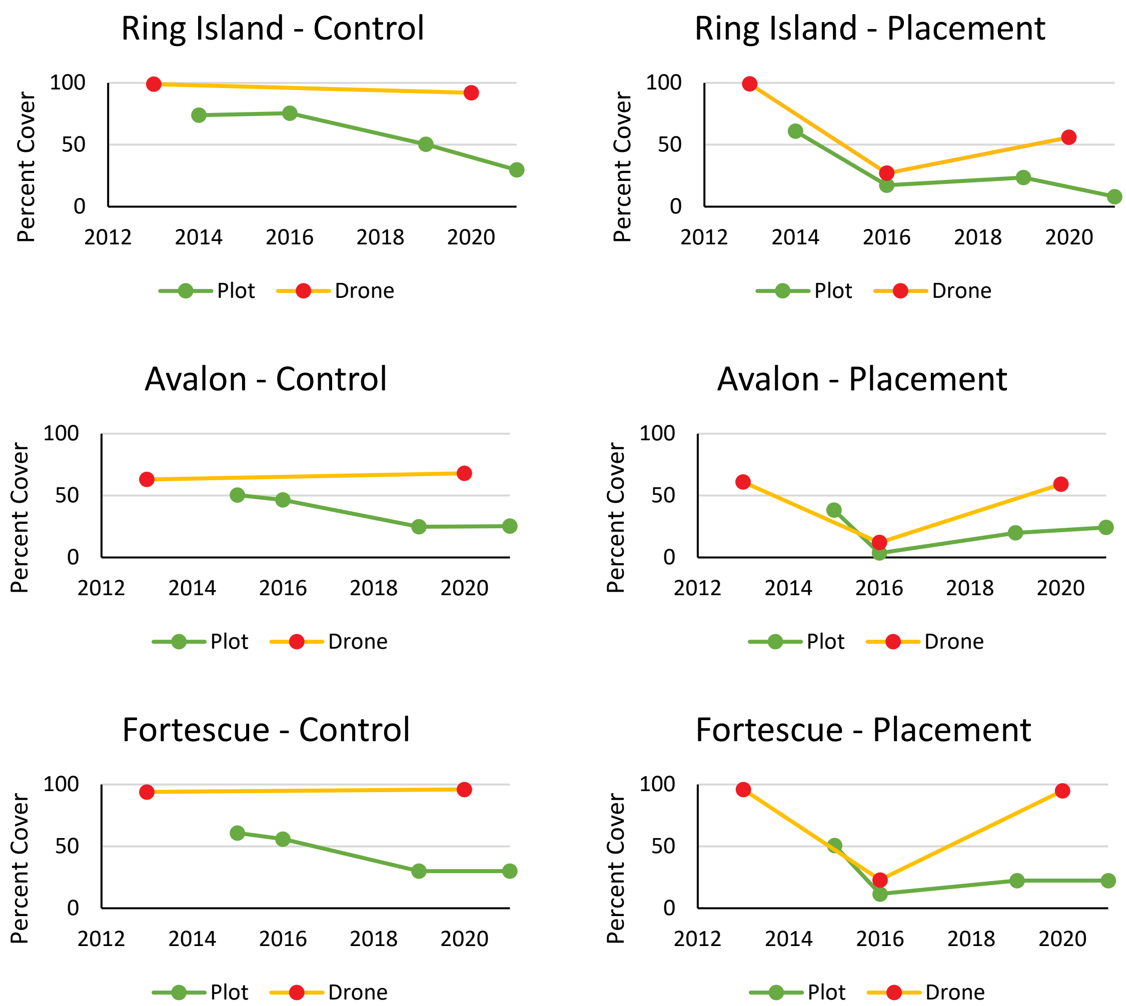


Figure 41. *Vegetation percent cover comparison between drone classification and on-the-ground permanent plot measurements. Drone imagery consistently was found to overestimate vegetation percent cover at a landscape level.*

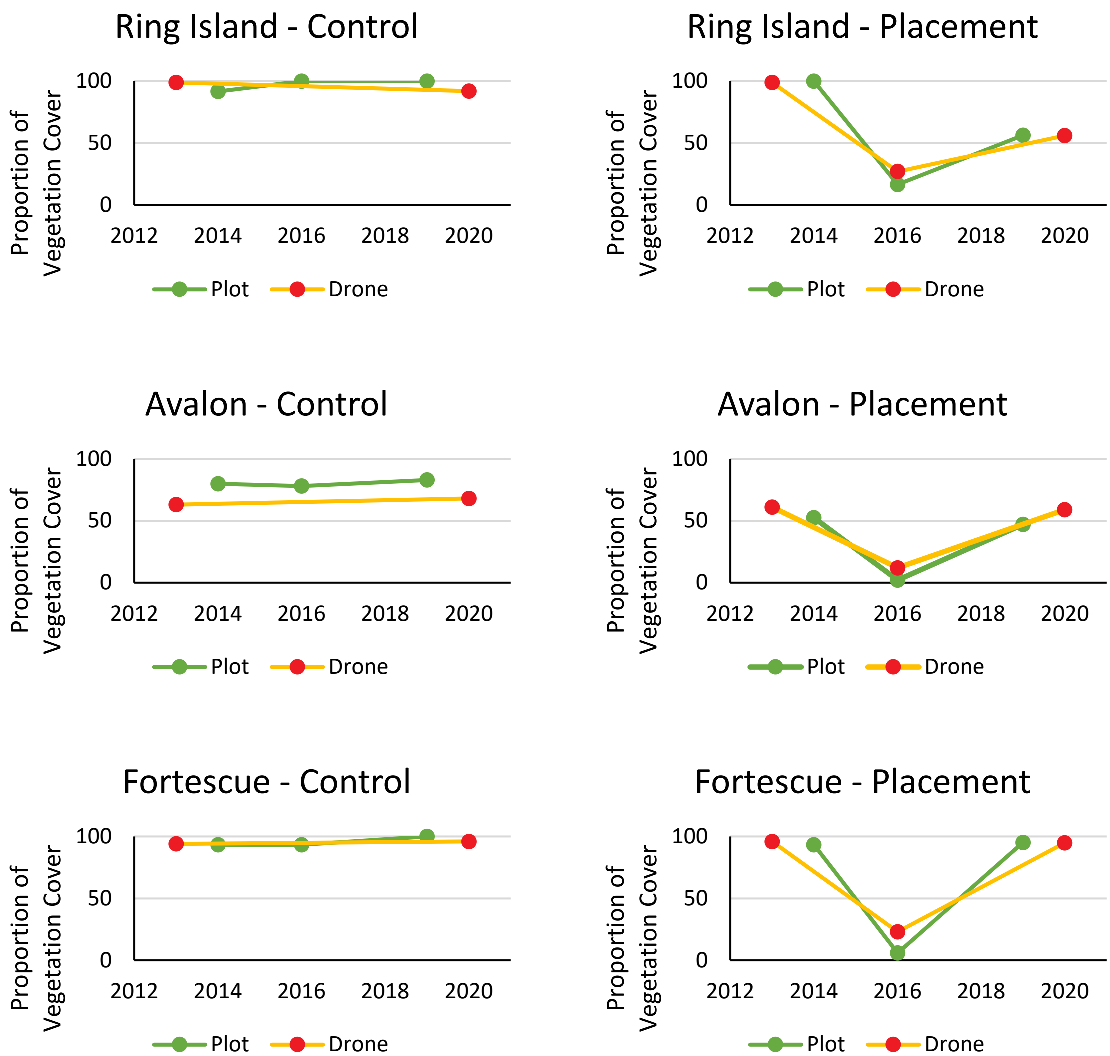


Figure 42. Habitat proportion comparisons between drone classification and on-the-ground permanent plot measurements. Drone imagery and plot measurements were closely matched as the two measurement types can determine habitat ratios more easily across a landscape level.

Conclusions

- Revegetation after placement of dredged material may take several years to reach baseline conditions. No sites reached baseline conditions by the time we collected the 2020 drone imagery five to six years post sediment addition. However, Avalon was the closest to reaching baseline vegetation conditions at 61.7% cover in 2020 compared to 67.1% cover in 2013. Ring Island revegetated up to 65.2% in 2020, with a baseline 2013 cover of 85.6%. Fortescue had the most vegetation cover by 2020 at 77.1%, with a baseline 2013 cover of 87.0%.

- Rates of revegetation are controlled by several factors, especially elevation of placed material, proximity to existing vegetation, thickness of material placed, erosion rates, and rates of ponding and subsidence. We found that vegetation tended to grow from outside of the placement area into the treated area, rather than from underlying surviving roots from previous vegetation.

- Drone imagery may overestimate vegetation cover due to the difficulty in analyzing density at a smaller scale compared to on-the-ground vegetation plots or it may provide a better estimate because plots were located away from the edge of where sediment was placed. Drone imagery was found to closely match the proportions of habitat cover, as this was a landscape-level metric.

- Areas of significant ponding may not be possible to fill in due to the weight of sediments and local subsidence rates. Although we filled in large ponds within Avalon, we quickly saw the ponds reemerge, although the area of the ponds was slightly decreased.

- Sediment addition can lend itself towards increasing subsidence, resulting in additional low-lying areas and ponding, as demonstrated by the Fortescue site. How sensitive a site is to subsidence and compaction relative to sea-level rise should be considered during design.

- Based on these results, we can address the following monitoring questions: 1) “Does the marsh ecosystem (i.e., structure and functions) recover or show uplift within two to three years of dredged material placement? (And the related “How long does it take for the marsh to recover and achieve uplift?”)” and 2) “Were there any unexpected outcomes of the placement of dredged material?”

  - 1) Revegetation after placement of dredged material may take more than six years to reach baseline conditions. No sites reached baseline conditions by the time we collected the 2020 drone imagery five to six years post sediment addition. However, Avalon was the closest to reaching baseline vegetation conditions at 61.7% cover in 2020 compared to 67.1% cover in 2013. Ring Island revegetated up to 65.2% in 2020, with a baseline 2013 cover of 85.6%. Fortescue had the most vegetation cover by 2020 at 77.1%, with a baseline 2013 cover of 87.0%. Post-placement monitoring timing should strive towards at least a decade due to the slow pace of recovery after sediment placement.

  - 2) Habitat classification revealed that sediment placement at the ENH resulted in a “bathtub” feature, where vegetation grew specifically at lower elevations and higher elevations, but not between. The exact cause of this ring of bare habitat is unknown, but it could be related to patterns of vegetation regrowth, where lower vegetation patches could be a result of higher seed counts from nearby vegetation patches and incoming flooding, and higher elevation vegetation patches could be a result of avian dispersal. Additionally, there could be underlying sediment changes that resulted in a gradient of prohibitive growing conditions.

7. Epifaunal Macroinvertebrates and Bioturbation

Adrianna Zito-Livingston[1] and Elizabeth Muntean[1]
[1]The Nature Conservancy

Epifaunal macroinvertebrates (EMI) (for example, mussels, snails, and crabs) provide important trophic linkages within salt marsh ecosystems. They are prey for a variety of avian and aquatic species, grazers of vegetation, and contribute to the structure of the marsh soil by digging burrows (*Uca* sp., fiddler crabs) or binding soils together, preventing erosion (*Geukensia demissa,* ribbed mussels; Bertness 1984). EMI are vulnerable to sudden changes in microhabitats within the marsh because of their relatively small size, and in some cases limited mobility. Monitoring EMI is important to document the effects of dredged sediment placement on faunal species diversity, especially in the case where a significant decline or an increase in species diversity or abundance could result in dramatic shifts in the ecological function of the marsh plain. the placement of dredged material.

Monitoring Design

Table 23. *Design and metrics used to monitor epifaunal macroinvertebrates at Ring Island, Avalon, and Fortescue project sites.*

Metrics	Method	Design
Species Richness	Presence or absence of species or crab boroughs	Surveys completed in ¼ m^2 section of permanent monitoring plots once during peak of growing season. BACI design.
Species Diversity		Surveys completed in ¼ m^2 section of permanent monitoring plots once during peak of growing season. BACI design.
Species Abundance		Surveys completed in in ¼ m^2 section of permanent monitoring plots once during peak of growing season. Post placement only.

EMI metrics (Table 23) were monitored by TNC in 0.25 m^2 subplots within the permanent 1 m^2 vegetation monitoring plots (see Section 5: Vegetation from Plot Based Monitoring) concurrent with annual vegetation sampling. Burrows were used as a proxy for crab abundance since crabs are highly mobile.

Paired Samples Wilcoxon tests were conducted on *Melampus bidentatus*, *Geukensia demissa*, and *Uca pugnax* burrow counts (Table 24) to compare abundances before placement (PRE) and the first year following placement (PP1) and to compare abundances before placement and in the final year post-placement (PP6/7). When the data were parametric, paired t-tests were used, as indicated.

Ring Island

At Ring Island, there were no *M. bidentatus* counted in plots before treatment (Table 24). The first year post-treatment averaged 1.6 *M. bidentatus* per plot (n = 9, Wilcox's $p > 0.05$, d = 0.33). By PP7 *M. bidentatus* averaged 9.1 per plot (n = 9, Wilcox's $p < 0.05$, d = 0.76), a significant increase from the mean of 0 in PRE. Due to missing data, we could not determine the averages of *G. demissa* and *Uca pugnax* in the pre-treatment year for Ring Island plots. However, the average number of *G. demissa* per plot in PP1 at Ring Island was 4.6 and 0.7 in PP7 (n = 12, Wilcox's $p > 0.05$, d = -0.36). *U. pugnax* burrows averaged 3.4 per plot in PP1 and 7.8 by PP7 (n = 12, Wilcox's $p > 0.05$, d = 0.41).

Table 24. *Average EMI counts per plot in the years PRE, PP1, and PP6 for Avalon and Fortescue and PP7 for Ring Island. Values highlighted in red and green had a difference in means compared to PRE averages that was significantly different than zero. Green indicates a difference in means significantly greater than zero (positive change in mean count) and red indicates a difference in means significantly less than zero (negative change in mean count).*

EMI Species	Year	Fortescue	Ring Island	Avalon
Melampus bidentatus	PRE	25.8	0	10.6
	PP1	0.7 ↓	1.6	0 ↓
	PP6/7	3.7 ↓	9.1 ↑	4.9 ↓
Geukensia demissa	PRE	2	-	-
	PP1	0	4.6	-
	PP6/7	0.4	0.7	-
***Uca pugnax* (burrows)**	PRE	10.3	-	2.3
	PP1	0.8 ↓	3.4	6.7 ↑
	PP6/7	2.7 ↓	7.8	10.0 ↑

Avalon

At Avalon, the mean number of *M. bidentatus* per plot was 10.6, and in PP1 the mean was 0 snails per plot. The average population of snails per plot in PP1 was significantly lower than in PRE (n = 18, Wilcox's $p < 0.05$, d = -0.44). The average number of *M. bidentatus* per plot in PP6 was 4.9, also a significantly lower average than in PRE (n = 17, Wilcox's $p < 0.05$, d = -0.289). Significance change in *G. demissa* was not detectable at Avalon due to only one out of 19 plots containing any mussels at either time point. The single plot that did contain mussels in PP1 increased from 2 individuals to 8 individuals in PP6. The mean number of burrows per plot increased significantly over time (Figure 43). Burrow count means increased from 2.3 in PRE to 6.7 in PP1 (n = 18, Wilcox's $p < 0.05$, d=0.58). The mean number of burrows per plot at Avalon in PP6 was 10.0, and the difference in mean to PRE counts was significantly greater than 0 (n = 14, paired t-test $p < 0.01$, d = 0.96).

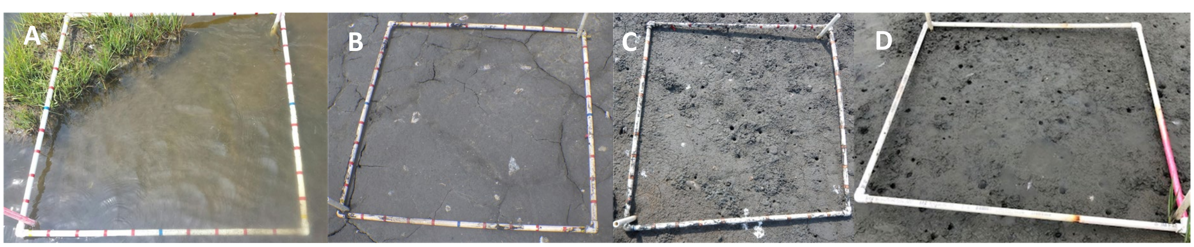


Figure 43. *Photo evidence of increase in burrow count during A) PRE, B) PP1, C) PP2, and D) PP3 in an Avalon Placement plot.*

Fortescue

At Fortescue, the mean number of *M. bidentatus* per plot in PRE was 25.8 and 0.7 in PP1, a significant decrease (n = 12, Wilcox's $p < 0.01$, d = -0.93) By PP6, the mean number of *M. bidentatus* per plot at

Fortescue was 3.7 (n = 12, Wilcox's $p < 0.05$, d = -0.78) the differences in means between PRE and PP6 still significantly greater than 0. The average number of *G. demissa* per plot was 2 in PRE and 0 in PP1, not a significant change in means (n = 12, Wilcox's $p > 0.05$, d = -0.49). By PP6 the mean per plot had increased to 0.4, not significantly different from PRE (n = 12, Wilcox's $p > 0.05$, d = -0.37). Crab burrows at Fortescue started with a mean of 10.3 burrows per plot in PRE and were reduced to 0.8 burrows per plot in PP1 (n = 12, Wilcox's $p < 0.05$, d = -1.23). By PP6 average burrow counts per plot had increased to 2.7, though still significantly different from PRE levels (n = 12, paired t-test $p < 0.05$, d = -0.92)

Conclusions

- The three most dominant EMI species found at all three project sites were salt marsh snails (*Melampus bidentatus*), ribbed mussels (*Geukensia demissa),* and burrowing crabs (fiddler crabs, *Uca* sp*).*

- The initial decline in *M. bidentatus* and *G. demissa* abundances may be attributed to direct burying by the placement of dredged material. *M. bidentatus* and *G. demissa* are also closely associated with vegetation, specifically *Spartina alterniflora,* so it is intuitive that as vegetation cover declined post-placement, so did the abundance of these species. On the contrary, crabs are highly mobile and were observed to burrow within the placed sediment. Ring Island and Avalon had higher abundances of crab burrows in the bare, non-vegetated placement areas than in the vegetated control areas post-placement. Burrows at each site were primarily made by *Uca sp*. (both *Uca pugnax* and *Uca minax were* observed) as opposed to *Sesarma reticulatum* (as indicated by their shape and size). Though mortality via burying is thought to have occurred initially within the footprint of the project, surviving *Uca sp.* from the surrounding marsh complex quickly recolonized the top layer of the dredged material at Avalon (but not at Fortescue). Although purple marsh crabs (*S. reticulatum*) were observed at all three sites, they were found in very low abundances, and a caging study was utilized to determine that herbivory by crabs was not a problem for recovering marsh vegetation in the study area.

- Based on these results, we can address the following monitoring question: "Does the marsh ecosystem (i.e., structure and functions) recover or show uplift within two to three years of dredged material placement? (And the related "How long does it take for the marsh to recover and achieve uplift?")"
    - Only *Uca pugnax* showed significant increases soon after placement and only within Avalon, and *Melampus bidentatus* took greater than six years to show significant recovery and only within Ring Island. Uplift of the epifaunal macroinvertebrates was not achieved soon after placement. While some species counts have recovered, it may be unlikely for all species to recover at all sites within a decade of the placement. Additional monitoring may be needed to fully determine the timescale in which these representative species will recover and exceed their previous population counts.

8. Penetration Resistance

Elizabeth Muntean[1], Jessie Buckner[1], Metthea Yepsen[1,2], Adrianna Zito-Livingston[1]
[1]The Nature Conservancy, [2]New Jersey Department of Environmental Protection

Bearing capacity measures the loading response or maximum force that the substrate can support and acts as an indicator of the firmness of the substrate. In traditional wetland monitoring, penetration resistance is used as a proxy for vegetative biomass belowground. The addition of dredged material changes the marsh substrate composition which influences penetration resistance and the ability of vegetation and EMI to colonize and grow. Harder substrates may be more difficult for roots and shoots to penetrate. Conversely, soft substrates may not be stable enough for root systems. By collecting penetration resistance data, vegetation success in relation to substrate hardness can be measured and optimal substrate hardness levels can be understood.

Monitoring Design

Table 25. *Design and metrics used by TNC for penetration resistance measurements at Ring Island, Avalon, and Fortescue projects.*

Design	Bearing Capacity & Sediment Characterization
Impact and Control Measured Once Annually During Peak Growing Season in 2016 and 2017 (July to September) next to permanent monitoring plots (see Section 5: Vegetation)	Penetration Depth Loading Response Depth of Standing Water

Penetration resistance metrics in Table 25 were measured by TNC directly outside a corner of the permanent vegetation plots (see Section 5: Vegetation from Plot Based Monitoring). A 2-inch diameter capped PVC tube was placed onto the wetland soil surface, and a standard force was applied with a slide hammer. The depth that that PVC tube sunk into the marsh after repeated drops of the slide hammer was used to calculate the penetration resistance. Soil color and texture were also documented.

Ring Island TLP

Because there was only one bare plot in the control area and no pool plots, we did not conduct a comparison of penetration resistance by habitat type at Ring Island (Figure 44). No significant differences were found comparing treatment and control plots (ANOVA: $p > 0.05$) or in treatment plots over time (Tukey; $p > 0.05$). However, there was an increase in penetration depths in the control plots from PP2 to PP7, although insignificant (Tukey; $p > 0.05$).

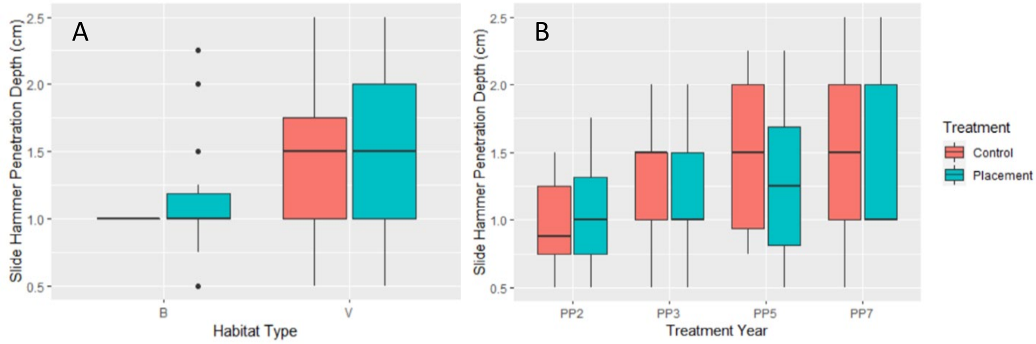


Figure 44. *Difference in penetration depth between Ring Island control and placement areas by A) habitat type, where B = bare and V = vegetated, and by B) years post-placement. Data shown includes median with interquartile ranges, minima, maxima, and outliers.*

Avalon

Soil penetration depth, which was only measured post-placement, was compared between control and treatment plots over time and between habitat types (Figures 45 and 46). Over the years, penetration depths have not significantly changed in control or treatment plots (Friedman; $p > 0.05$ and ANOVA; $p > 0.05$ respectively). Within control areas, soil penetration depth was significantly lower in vegetated plots compared to either bare ($p < 0.001$) or pool ($p < 0.01$) plots and was similar between bare and pool plots ($p > 0.05$). Penetration depths were lower in bare treatment plots than bare control plots (ANOVA; $p < 0.01$) most likely due to the compaction of dredge material during placement compared to the natural peat substrate of the controls.

For Avalon plots that started as pools, penetration depths were higher than in plots that started as platform (ANOVA; $p < 0.001$). Penetration depth was also significantly different between habitat types in these plots (ANOVA; $p < 0.01$) with penetration depths being lowest in bare plots and highest in pool plots.

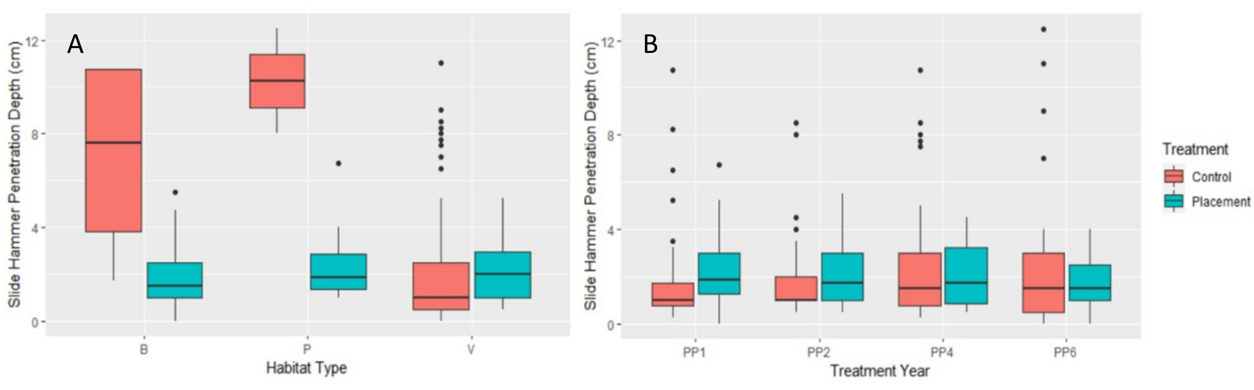


Figure 45. *Difference in penetration depth of Avalon control and placement platform plots by A) habitat type, where B = bare, P = pool, and V = vegetated, and by B) years post-placement. Data shown includes median with interquartile ranges, minima, maxima, and outliers.*

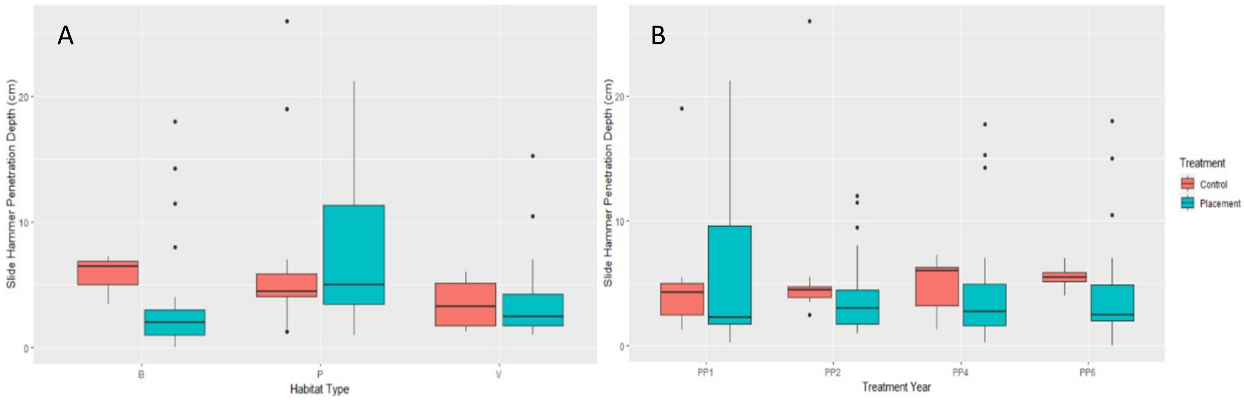


Figure 46. *Difference in penetration depth of control and placement pool plots by A) habitat type, where B = bare, P = pool, and V = vegetated, and by B) years post-placement. Data shown includes median with interquartile ranges, minima, maxima, and outliers.*

Fortescue

For Fortescue, soil penetration depths were log-transformed for normality. Penetration resistance was significantly affected by treatment (control vs. placement) (ANOVA, $p < 0.05$), but not by timeframe (ANOVA, $p > 0.05$). Placement plots had lower penetration depths than control plots every year that data was collected and for each habitat type (Figure 47). There were no pool plots in the control area, so pools are not part of the comparison by habitat type.

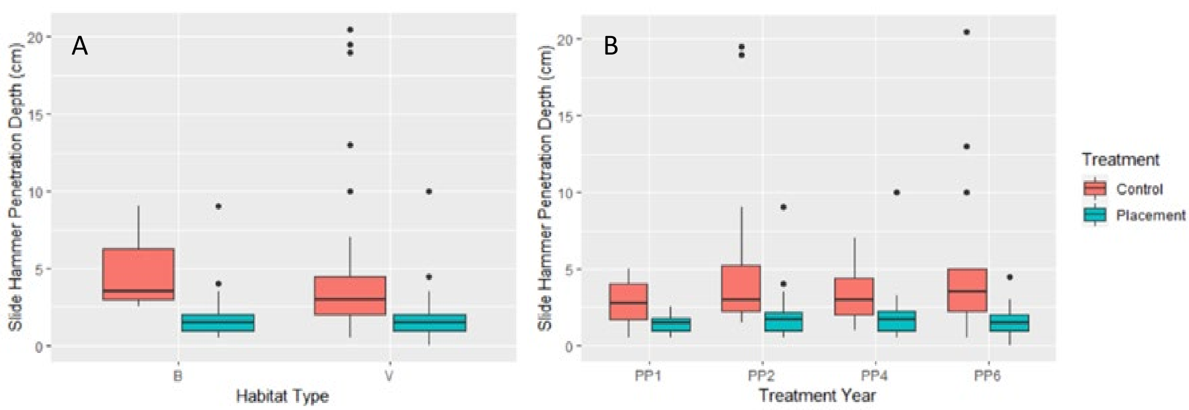


Figure 47. *Difference in penetration depth of control and placement plots by A) habitat type, where B = bare and V = vegetated, and by B) years post-placement. Data shown includes median with interquartile ranges, minima, maxima, and outliers.*

Conclusions

- Vegetated control plots at all sites had generally well-established root mats, resulting in relatively higher penetration depths than in bare control plots.
- At Avalon and Fortescue, it was observed that the surface sediments of bare placement plots were often dried and/or compacted, sometimes with a leathery algal mat on the surface that resisted penetration. It was hypothesized that compacted soils or thick algal mats might make it more difficult for plants to re-establish.

- In Avalon, penetration depths were lower in bare treatment plots than bare control plots, likely resulting from the compaction of dredge material during placement compared to the natural peat substrate of the controls.
- At Fortescue, placement areas had lower penetration depths than controls across time. Additionally, vegetated plots in placement and control areas had similar penetration depths. This could indicate that similar bearing capacity facilitated vegetation recovery within placement plots.

9. Benthic Infauna and Sediment Properties

Gary L. Taghon[1] and Rosemarie F. Petrecca[1]
[1]Rutgers, the State University of New Jersey

Benthic infauna live in the sediment and are prey items for juvenile fish and larger invertebrates that live in the marsh. Infauna play a role analogous to earthworms in keeping the sediment aerated, breaking down organic matter, and returning nutrients to the water column. Due to the intimate association of benthic infauna with the sediment in which they live, the placement of dredged material can have an immediate effect on the abundance and composition of the benthic community. Dredged material is likely to differ from natural marsh sediment in particle size, concentrations of key nutrients (carbon, nitrogen, sulfur, and phosphorus), and organic matter content – properties that fundamentally affect benthic infauna.

Table 26. *Design and metrics used for benthic infauna monitoring and sediment properties at all three project sites.*

Metrics	Method	Design
Benthic Infauna Species abundance Species diversity	3.8 cm diameter × 3 cm deep sediment cores sorted in lab	Ten samples collected at each marsh placement and control site during each sampling event. There were five sampling events in 2015 (Avalon and Ring Island only) and in 2016, two sampling events in 2017, and one in 2019. No baseline data were collected. Samples were collected in the same small area each time, except in 2019 when two separate placement sites were sampled. The placement site at Avalon received sediment in winter 2014–2015 and in winter 2015–2016.
Sediment Total organic matter Total C, N, S, P Grain size (phi)	7.6 cm diameter × 2 cm deep sediment cores processed in lab	Two samples collected at each marsh placement and control site during each sampling event. There were five sampling events in 2015 (Avalon and Ring Island only) and in 2016, two sampling events in 2017, and one in 2019. Samples from 2019 were not analyzed for P content due to Covid-19 restrictions. No baseline data were collected. Samples were collected in the same small area each time, except in 2019 when two separate placement sites were sampled. The placement site at Avalon received sediment in winter 2014–2015 and in winter 2015–2016.

Monitoring Design

The benthic infauna and associated sediment properties listed in Table 26 were monitored by Rutgers University. Ten benthic infauna samples and two soil samples were collected at each marsh placement and control site during each sampling event. There were five sample events in 2015 (Avalon and Ring

Island only) and 2016, two sample events in 2017, and one in 2019. Sample locations were in the same dredged material placement area at each project site, over time and were co-located with a vegetation plot. Given the heterogeneous nature of Avalon and Fortescue, the samples are not necessarily representative of the entire project site. This is especially true at Avalon, where conditions were the most heterogeneous. All of the samples were collected after the dredged material had been placed so only placement and control sites can be compared.

Results

Benthic Infauna

Ring Island

19 taxa were collected in the control area over all four years. Diversity was low; four taxa accounted for 97% of all individuals (Figure 48). There were little to no signs of recovery or recolonization of the placement area from 2015 through 2017. In 2019, the abundance of the opportunistic polychaete *Capitella* in one of the placement areas was similar to the control area.

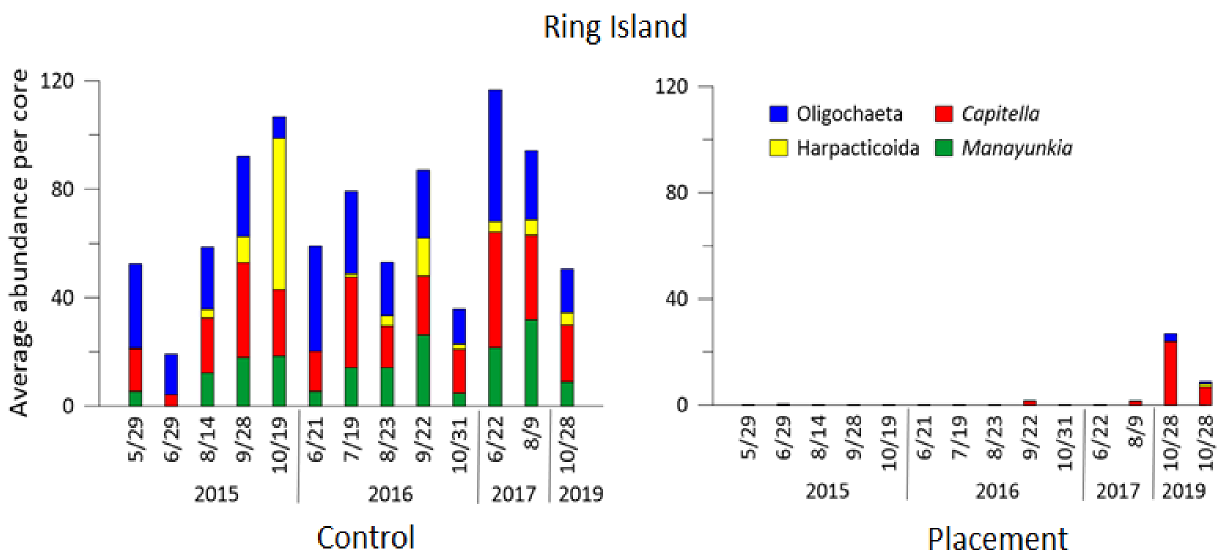


Figure 48. *Average abundance per core of the dominant benthic infauna taxa in control and placement areas at the Ring Island thin-layer project site. All samples were collected after placement occurred in 2014. The duplicate dates in the 2019 Placement bargraphs represent the two Placement areas that were sampled.*

Avalon Phase 2

21 taxa were collected in the control area over all four years. Diversity was low; five taxa accounted for 96% of all individuals (Figure 49). The Avalon "placement" samples were taken from Area C. This area had initially received dredged material in 2015 and again in 2016; therefore, the 2016 data is categorized as "one-year post-placement". The abundances of benthic infauna were minimal in 2015 and through September 2016. In October 2016, *Harpacticoida* began colonizing the area, with a mean abundance greater than those in the control area. The colonization by these small benthic copepods was coincident

with the appearance of *Capitella.* This may indicate facilitation of settlement by *Harpacticoida* by *Capitella*. These worms bind sediment particles with mucus to construct a 'tube' they reside within, and the copepods were found in large numbers within the mucus-sediment matrix.

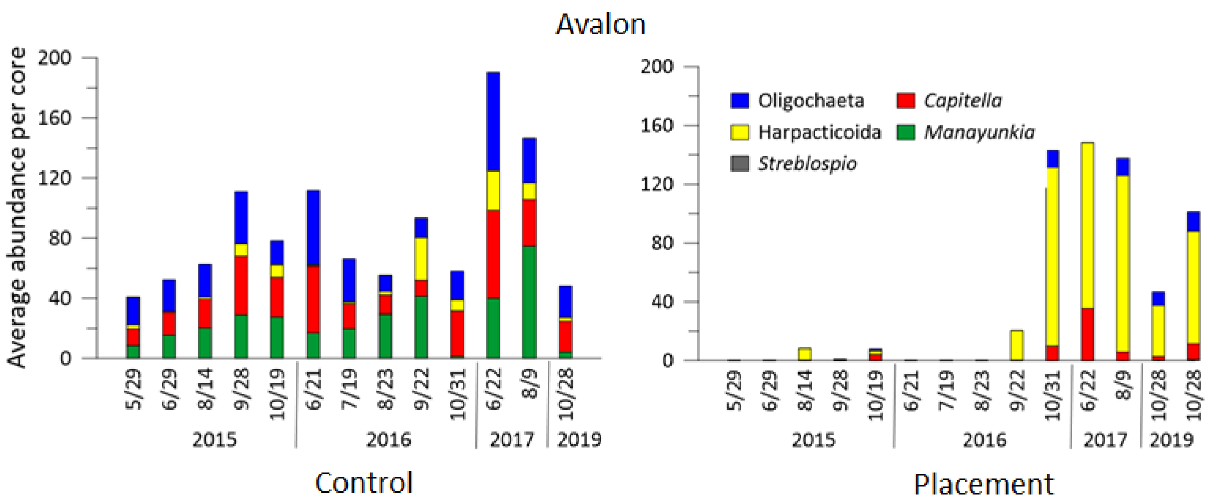


Figure 49. *Average abundance per core of dominant taxa in control and placement areas at the Avalon project site. The placement area sampled received sediment in the winter of 2014-2015 and winter 2015-2016. The duplicate dates in the 2019 Placement bargraphs represent the two Placement areas that were sampled.*

Fortescue

17 taxa were collected in the control area over all three years. Diversity was low; five taxa accounted for 96% of all individuals (Figure 50). There were no signs of recovery or recolonization in the placement area one-year post-placement (2016). Two years post-placement (2017) *Capitella* and *Harpacticoida* were found in the sampled placement area in numbers that were greater than those in the control area in the June sample. *Oliogochaeta* were also present but in lower abundance than in the control area. Numbers declined in the

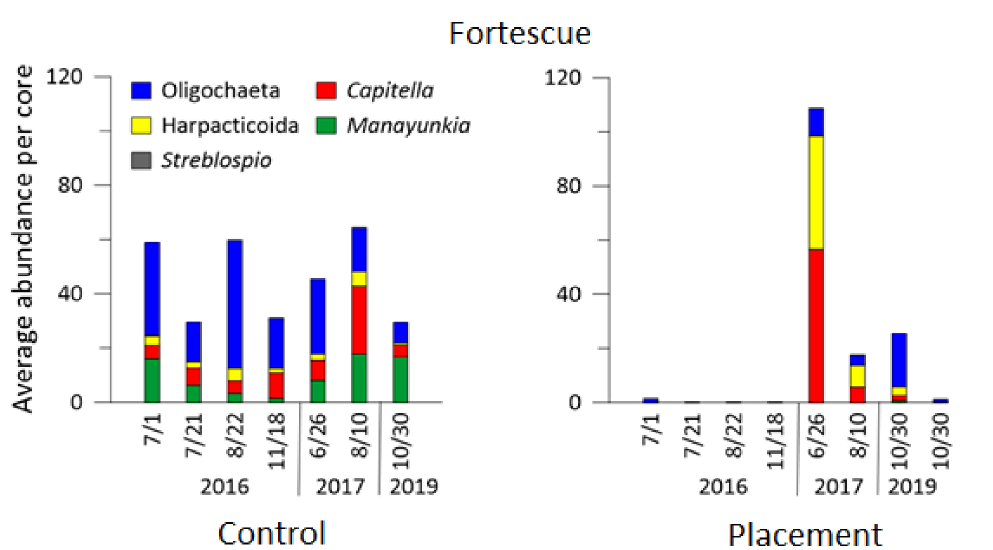


Figure 50. *Average abundance of benthic infauna per core of dominant taxa in control and placement areas at the Fortescue project site. All samples were collected after sediment was added in in the winter of 2015-2016. The duplicate dates in the 2019 Placement bargraphs represent the two Placement areas that were sampled.*

August samples, but all three taxa were present. In 2019 there were few infauna in one of the placement areas while the total abundance in the second placement area was similar to the control.

Sediment Properties

In the soils associated with the benthic infauna cores, soils in all marsh control areas were approximately 90% silt and clay (Figure 51). It was expected that placement areas had similar baseline conditions. In the soils associated with the benthic infauna cores, post-placement sediment had a much lower silt and clay content. Ring Island placement sediment was dominated by fine sand. Placement sediment at Avalon had 80% silt and clay when sampled in the summer and fall of 2015, but in 2016 and 2017 very fine sand and fine sand were the dominant size fractions. The area where samples were collected at Avalon received sediment additions in the winters of 2014 and 2015. Sediment was dredged from different parts of the navigation channel from one year to the next which is likely why the sand content of the sediment increased from 2015 to 2016. In 2019, Avalon placement sediment was again dominated by silt and clay. Fortescue placement area sediment had greater medium and coarse sand content than the other marsh locations.

Sediment sample sizes were low and the spatial distribution of the samples was limited to the same general location each time. This likely means that there was greater variability than is captured here. For

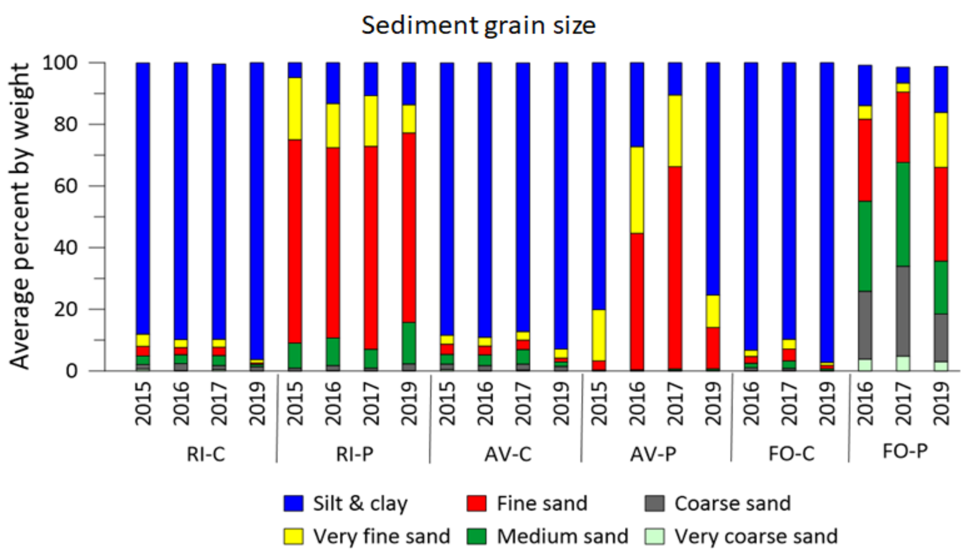


Figure 51. *Sediment size distributions at Ring Island (RI), Avalon (AV), and Fortescue (FO) post-placement in Control (C) and Placement (P) areas.*

example, based on navigation samples collected before dredging and a separate soils study conducted by Princeton Hydro at Avalon, there was a higher silt content in placed sediments than is reflected by the data. As a result, the sediment data collected at Avalon is not representative of the larger site and should only be used to interpret the associated benthic infauna data.

In sediment samples associated with benthic infauna cores collected post-placement, total organic matter, total carbon, total nitrogen, and total phosphorus concentrations were all much lower in placed sediments than in the controls (Figure 52). Sulfur concentration in the Avalon placement site sediment was similar to the control site. Ring Island placed sediment had lower sulfur concentration than the control

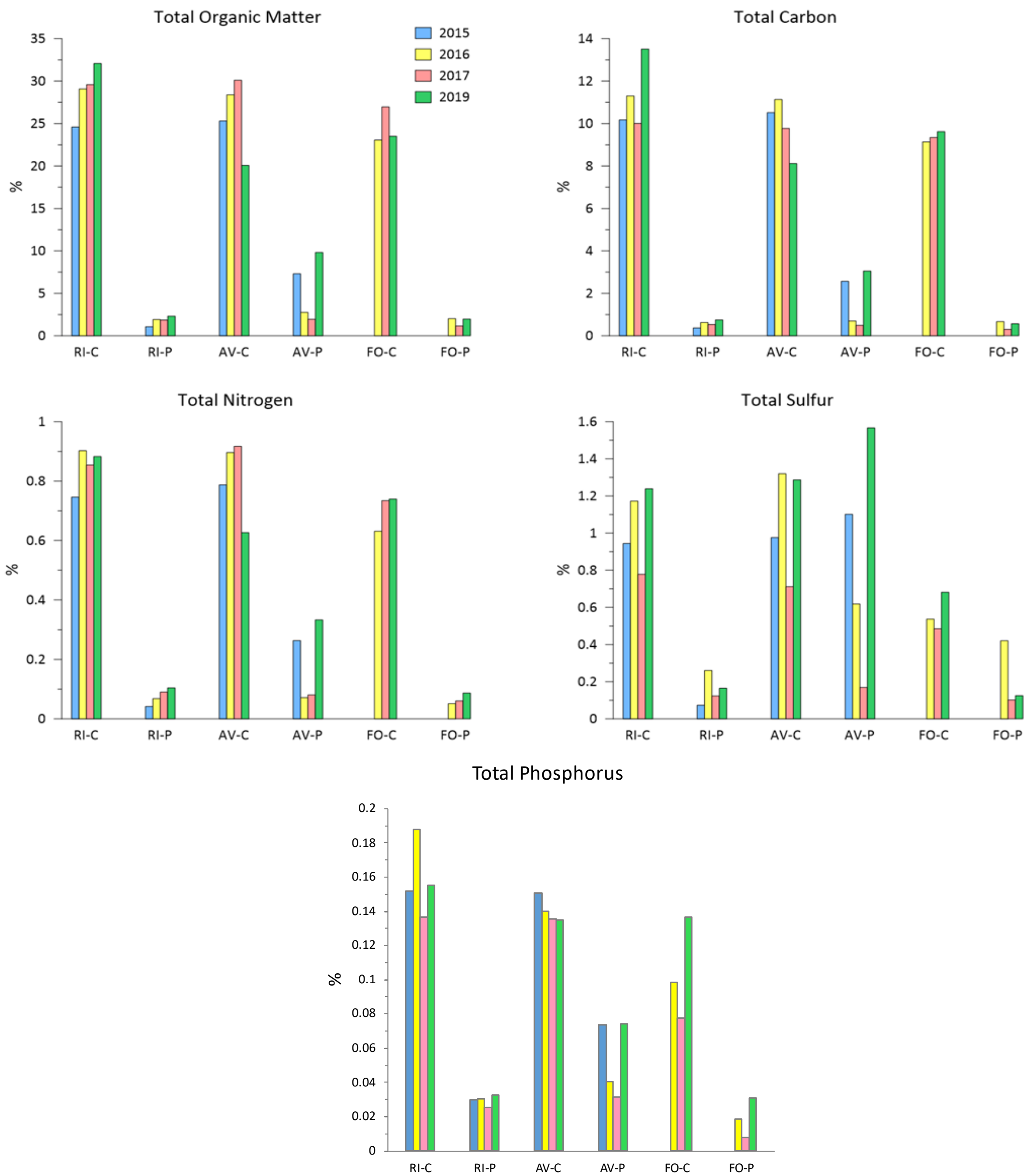


__Figure 52.__ Total Organic Matter, Carbon, Nitrogen, Sulfur, and Phosphorus concentrations of sediments at Ring Island (RI), Avalon (AV), and Fortescue (FO) post-placement in Control (C) and Placement (P) areas.

in all years. The sulfur concentration of placed sediment at Fortescue was similar to control sediment in 2016, then decreased substantially in 2017 and 2019.

Conclusions

- There was an immediate decline in the abundance of all benthic infauna across all three sites, most likely attributed to mortality via direct burial. The Ring Island site showed minimal signs of recovery of any taxa four years post-placement. In 2019, the abundance of the opportunistic polychaete *Capitella* in one of the placement areas was similar to the control area. Taxa well suited to disturbed environments began to recolonize the sediment at the Avalon site in August one-year post-placement, and at the Fortescue site in June two-years post-placement. While abundance was similar between placement and control sites at Avalon and Fortescue in 2019, the proportion of taxa making up the cores remained different.
- Nutrient levels were generally lower in all placement sites compared to controls, except in Avalon, where sulfur concentrations tended to be higher in 2015 and 2019 samples from placement areas.
- Differences in the recovery response of the benthic infauna community may be attributed to differences among the placement sites (acreage of placement, time of construction, elevation, grain size, and composition of the material, etc.).
- It was expected that the placed sediments would bring nutrients into the marsh. However, based on a limited sample size from sandy sediments, this was not the case. One potential reason for this may be the coarse grain size of the sediment samples. Sands contain less nutrients and contaminants than silts as they consist of little to no organic matter.
- A larger sample size and broader spatial distribution of samples are recommended for future projects.
- Based on these results, we can address the following monitoring questions: 1) "Does the marsh ecosystem (i.e., structure and functions) recover or show uplift within two to three years of dredged material placement? (And the related "How long does it take for the marsh to recover and achieve uplift?")" and 2) "Did differences in structural factors (e.g., grain size, elevation, placement thickness) correlate with recovery and uplift?"
    - 1) Ring Island surveys showed only minor recovery in population counts after five years post-placement. Avalon did have significant recovery in benthic infauna counts four years post-placement, nearly matching total counts in the control areas. However, there was a large shift in diversity, with fewer taxa represented post-placement. Fortescue had mixed results, with some surveys demonstrating high population counts and others with very low counts post-placement. There were additional diversity shifts in Fortescue populations post-placement, with fewer taxa represented in treated areas. For Ring Island and Fortescue, it may take several more years for populations to match and exceed previous conditions. Avalon, however, seems to have been successful in recovering its benthic infauna populations.
    - 2) All nutrient levels, except sulfur concentrations in Avalon, were decreased in placement

areas. This exception of sulfur in Avalon may be a potentially important parameter in benthic infauna recovery. This relates to the importance of sediment grain size and composition. Smaller grain-sized sediments are likely to impart more nutrients than sandier sediments, which may be a significant factor in the speed of marsh recovery after placement.

10. Avian Monitoring

Samantha Collins[1], Brittany Morey[1], Lisa Ferguson[1], Allison Anholt [1], and Princeton Hydro
[1]The Wetlands Institute

Coastal salt marshes provide habitat for many species of birds during nesting and migration with abundant food resources and low levels of human disturbance and mammalian predators. Avian species on the State and Federal Threatened and Endangered Species Lists use these areas for foraging, resting, and nesting. Some listed species documented in this project include black rails (*Laterallus jamaicensis*), black skimmers (*Rynchops niger*), red knots (*Calidris canutus*), and least terns (*Sternula antillarum*).

Monitoring Design

Avian surveys were performed seasonally at all three project sites in dredged material placement and control areas (Table 27) to assess the impacts of using dredged material for enhancement purposes on the avian community. Monitoring was conducted by The Wetlands Institute and Princeton Hydro. At Ring Island only, additional wildlife use surveys of the thin layer placement (TLP) areas and the elevated nesting habitat (ENH) were performed weekly from April through August. All final survey locations were chosen in the field based on accessibility and within major habitat types: low marsh, high marsh, pools, pannes, and control, and placement areas.

Table 27. *Metrics, method, and design for avian monitoring at all three project sites. The Before-After Control-Impact (BACI) design was utilized at Avalon (AV) and Fortescue (FT); 'Before' surveys were not performed at Ring Island (RI) due to timing restrictions.*

Metrics	Method	Design
Species Richness (SCj) Species Abundance Species Composition	Point Count and Playback-Response Surveys (Spring, Summer, Fall; AV, FT, RI)	B-A-C-I (AV & FT): Four placement and three control points were surveyed one time per season 2015-2017 Control-Impact (RI): Two placement and eight control points were surveyed two times per season 2015-2017; Four of these points (two placement and two control) were surveyed two times per season 2018-2019
Nest Status Predation Risk Number Location Behavior of Birds	Weekly surveys (April-August; RI)	Wildlife Use and Nesting Success Surveys (RI): Conducted at placement and control areas 2015-2019

The goals of the surveys were to (1) document which bird species use the dredged material placement areas during Spring, Summer, and Fall seasons in comparison to control areas and surrounding marsh, and (2) document how species of birds and other wildlife use the placement areas, and the marsh as a whole, during the breeding season.

Results

Ring Island

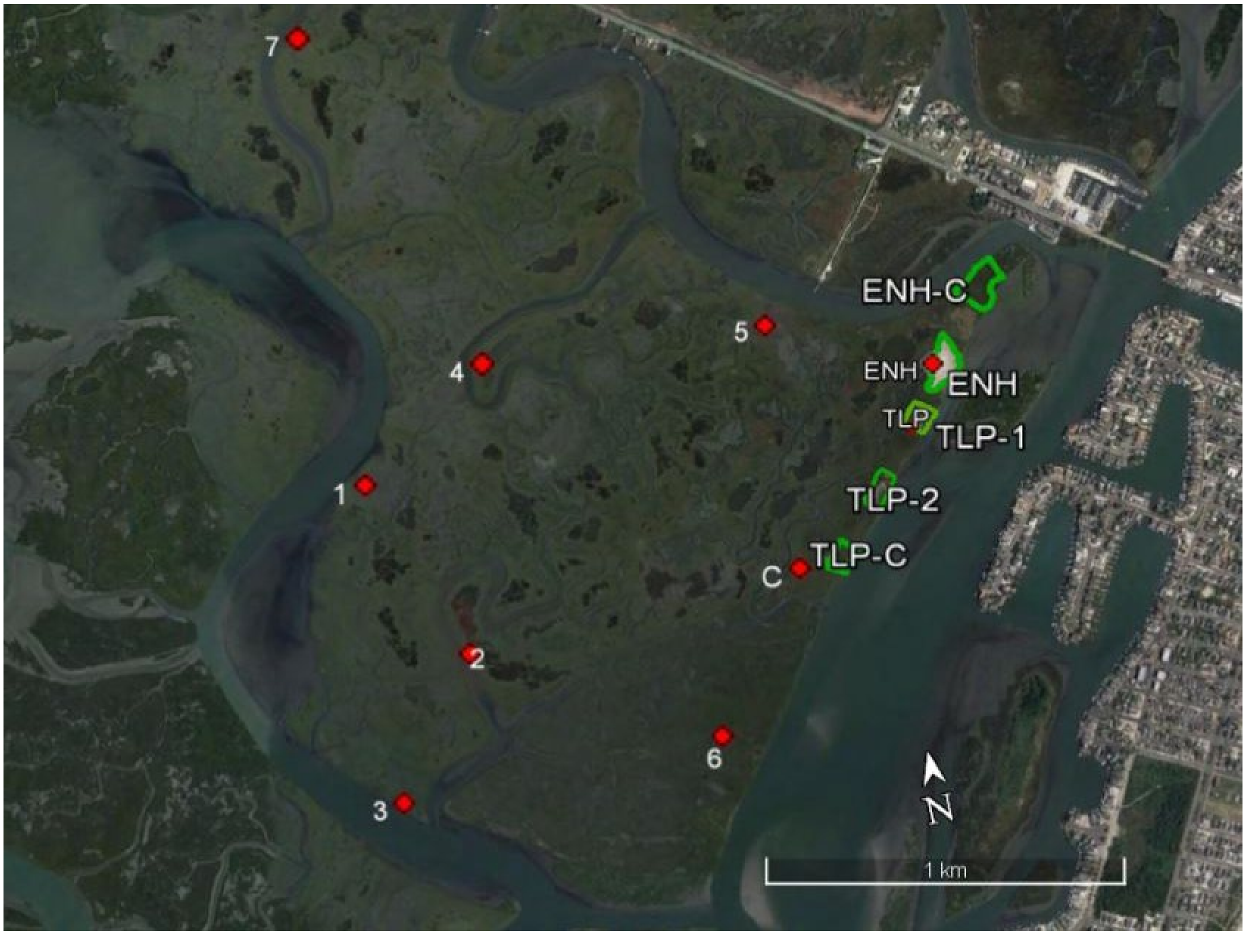


Figure 53. *Ring Island point-count locations for avian surveys indicated by red dots (N= 10; 2015-2017): Thin Layer Placement site (TLP), Elevated Nesting Habitat (ENH), and eight control points (C, 1-7). Four of these points (TLP, ENH, C, and 5) were surveyed in 2018-2019. Areas surveyed for wildlife use and nesting (2015-2019) indicated by green polygons: Thin Layer Placement areas (TLP 1 & 2), Elevated Nesting Habitat (ENH), and two Control sites (TLP-C and ENH-C). TLP point-count location is located within TLP-1.*

All surveys at Ring Island were conducted post-construction so placement areas (ENH and TLP areas) were compared to control areas, shown in Figure 53. A total of 72 species from 8 guilds were documented during point-count surveys (Guild and number of species: Gull: 4; Passerine: 23; Piscivore: 8; Rail: 1; Raptor: 5; Shorebird: 18; Wader: 8; Waterfowl: 5; Figure 54). The most abundant species was the Laughing Gull (*Leucophaeus atricilla*), which nests in large numbers on Ring Island and was documented at placement and control points. Clapper Rail (*Rallus crepitans*) was the only Rail species documented. Figure 55 shows the mean seasonal species richness at points surveyed in 2015-2019. A subset of the ten original points surveyed in 2015-2017 was included for continued monitoring (points TLP, ENH, C, 5; 2018-2019).

Species richness was affected by point, season, year, and their two-way interactions in a least squares means analysis of the 10 points surveyed 2015-2017 ($F = 10.9$, $df = 17,72$, $p < 0.01$; Figure 55). Species

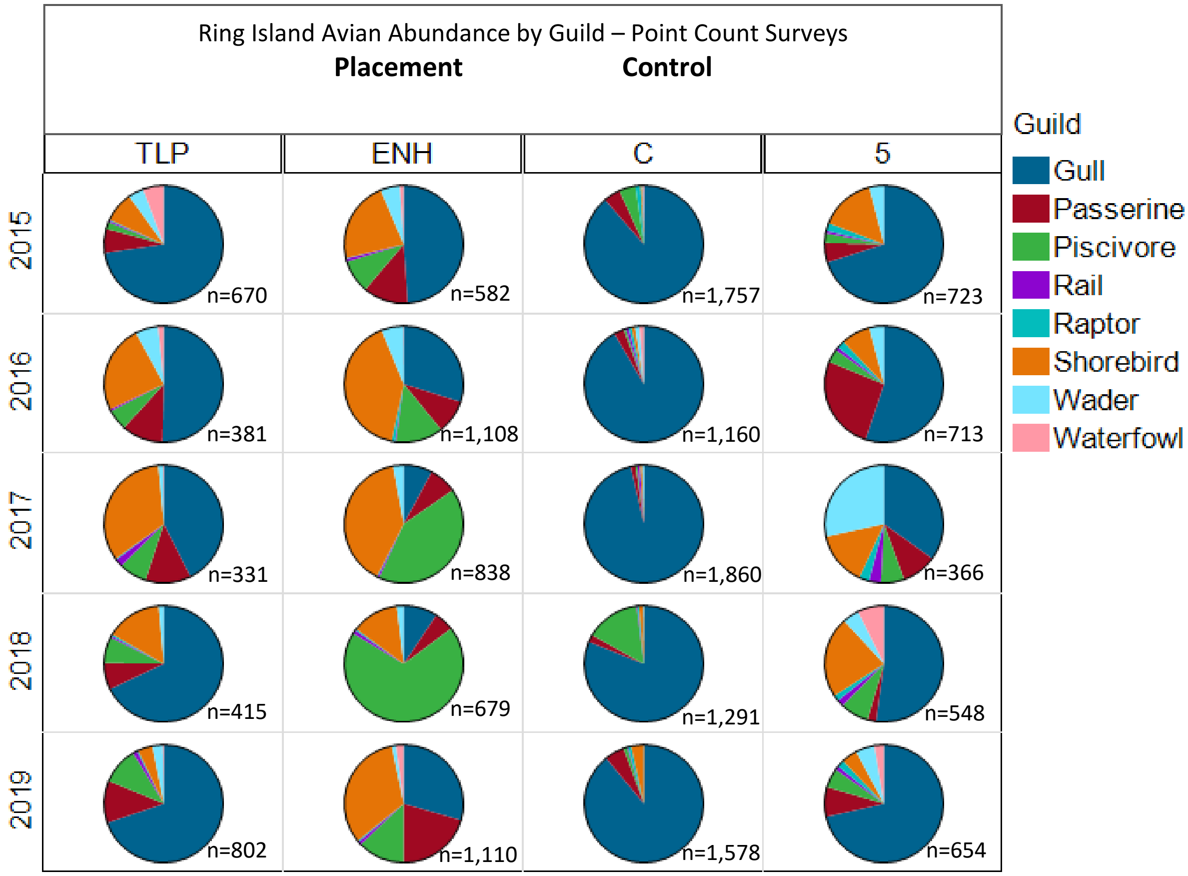


Figure 54. *Avian abundance by guild at Ring Island placement points (left) and control points (right) 2015-2019; n =total birds counted during surveys.*

richness varied by survey point (F = 18.8, df = 9, $p < 0.01$), with a greater number of species using the ENH area compared to other points. TLP fell in a mid-range of species richness with five control points. Seasonal and annual effects interacted (F = 3.4, df = 4, $p < 0.05$), which was due in part to high annual variability during fall surveys. No other effects were significant (all $p > 0.05$). These relationships were examined in a separate least squares means analysis of the four points surveyed over five years and similar results emerged (F = 8.5, df = 17,42, $p < 0.01$). Species richness varied by point (F = 35.1, df = 3, $p < 0.01$), and was highest at ENH, fell in the mid-tier at TLP and control point 5, and was lowest at control point C. Year and season had a significant interactive effect (F = 2.8, df = 8, $p < 0.05$). The proportion of birds belonging to each guild varied among the four points, shown in Figure 54. The high abundance and low species richness of point C are due to its proximity to high-density nesting area for Laughing Gull. The TLP point also had a large proportion of gulls but was used by a greater proportion of species belonging to other guilds, including 13 species of shorebirds. Species richness and the guild composition of species using the TLP was most similar to point 5, which is located near a panne that attracted a variety of species including 13 shorebird species. The guild composition of the ENH point is reflective of high species richness at the point, and a large proportion of piscivores (Black Skimmers and terns species) nest on the site in most years following construction. The ENH point was also used by a diversity of species during spring and fall migration and had the greatest diversity of shorebirds, at 16 species.

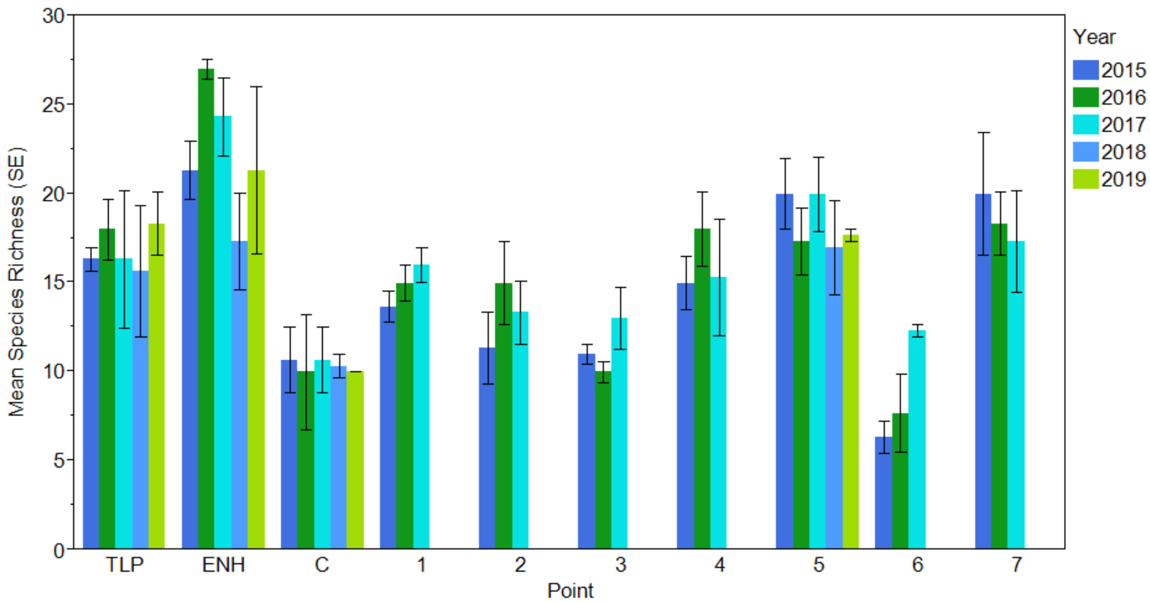


Figure 55. *Mean species richness counted within 100m at 10 observation points during point-count avian surveys at Ring Island, 2015-2019. Two surveys were conducted per season (Spring, Summer, Fall). Only four points were surveyed in all years. Error bars represent one standard error.*

Table 28 shows the number of breeding pairs, nests, and reproductive metrics documented during weekly surveys from April - August 2015-2019. Nests of nine species were documented in the study areas and a portion was monitored weekly. Black Skimmers nested on the ENH site beginning the third year following construction, and tern species nested on the site in all years of the study. Only two species were confirmed nesting on the ENH in 2019, the fewest since construction, while the number of species nesting in the TLP areas was the highest over five years. Predation of nests by Eastern Red Fox (*Vulpes vulpes*; 2017 and 2019) and Fish Crows (*Corvus ossifragus*; 2016-2019) reduced reproductive outcomes at placement and control sites. Several monitored nests were lost to flooding over the monitoring period. These nests were located in control and TLP areas or in areas of low elevation on the ENH (*i.e.*, not the nesting platform). The establishment of mixed species of vegetation on the ENH nesting platform reduced habitat quality beginning in 2018 and was adaptively managed with fire, sediment placement, mechanical and hand removal, and saltwater solution. The ENH provided nesting habitat for diamondback terrapins *Malaclemys terrapin*) and spawning horseshoe crabs *(Limulus polyphemus*).

Table 28. *Number of breeding pairs, nests, apparent hatch success (% nests that hatched ≥1 egg), and productivity (number of chicks fledged/pair) from Elevated Nesting Habitat (ENH), Thin Layer Placement (TLP-1 and TLP-2), and control areas (TLP-C and ENH-C) study areas on Ring Island, 2015-2019. Metrics for colonial nesting species (Black Skimmer, Laughing Gull, Common Tern (Sterna hirundo), and Least Tern (Sternula antillarum)) represent minimum estimates. Unk=Unknown; NE=Not Evaluated.*

Metric	**Breeding Pairs**					**Nests**					**Hatch Success (%)**					**Productivity**				
Year	2015	2016	2017	2018	2019	2015	2016	2017	2018	2019	2015	2016	2017	2018	2019	2015	2016	2017	2018	2019
ELEVATED NESTING HABITAT																				
Black Skimmer	0	0	37	12	10	-	-	51	13	0	-	-	25.5	23.0	-	-	-	0.1	0	-
Common Tern	0	0	85	200	110	-	-	100	200	132	-	-	32.0	42.5	0	-	-	0.6	0.2	0
Least Tern	2	16	24	4	0	2	16	33	6	-	100	56.3	57.6	0	-	1.5	0.3	1.3	0	-
Great Black-backed Gull	1	1	1	0	0	1	1	2	-	-	100	100	0	-	-	2.0	1.0	0	-	-
American Oystercatcher	2	1	4	3	2	3	2	6	3	0	67	50	50	100	-	2.5	2.0	1.5	0.7	-
Willet	0	1	2	1	3	-	1	2	1	3	-	100	0	100	0	-	NE	0	NE	0
Clapper Rail	0	1	0	0	0	-	1	-	-	-	-	100	-	-	-	-	NE	-	-	-
Seaside Sparrow	0	0	1	0	0	-	-	1	-	-	-	-	0	-	-	-	-	0	-	-
THIN LAYER PLACEMENT																				
Laughing Gull	5	9	0	0	10	5	9	-	-	9	NE	100	-	-	22.0	NE	NE	-	-	0
American Oystercatcher	0	0	1	0	1	-	-	1	-	2	-	-	0	-	0	-	-	0	-	0
Clapper Rail	2	5	0	0	5	2	5	-	-	5	50.0	80.0	-	-	60.0	NE	NE	-	-	NE
Seaside Sparrow	0	0	0	0	2	-	-	-	-	2	-	-	-	-	0	-	-	-	-	0
CONTROL AREAS																				
Great Black-backed Gull	0	0	0	1	0	-	-	-	1	-	-	-	-	Unk	-	-	-	-	NE	-
Laughing Gull	0	20*	175	37*	125	-	20*	175	37*	150	-	85.0	85.7	54.0	27	-	NE	1.8	NE	0.2
American Oystercatcher	1	0	0	1	0	1	-	-	1	-	0	-	-	0	-	0	-	-	0	-
Clapper Rail	2	1	2	4	4	2	1	2	4	4	50.0	Unk	Unk	Unk	Unk	NE	NE	NE	NE	NE
*Only includes nests monitored, not total number of nests and breeding pairs in area																				

Avalon

In 2015, before the placement of dredged material, species richness varied by survey point, with placement areas exhibiting similar to or greater richness than the control areas (Figure 56). In 2016, 1-year post-placement, richness declined at all control survey points and generally increased at all placement survey points. Richness increased within the control areas in 2017, two-years post-placement, whereas richness generally decreased within the placement areas.

In the control areas, species richness increased significantly between 2016 and 2017 ($p < 0.05$); however, no significant change in species richness was detected in the placement areas between 2015, 2016, and 2017. When comparing control areas to placement areas, species richness was significantly higher ($p <$

0.05) in the placement areas in 2016 (one-year post-placement).

Fortescue

Figure 57 presents avian species richness at Fortescue in placement and control areas pre-placement (2015) and post-placement (2016 and 2017). No statistically significant change in avian species richness was detected within control or placement areas throughout all three years, or between control and placement areas throughout all three years.

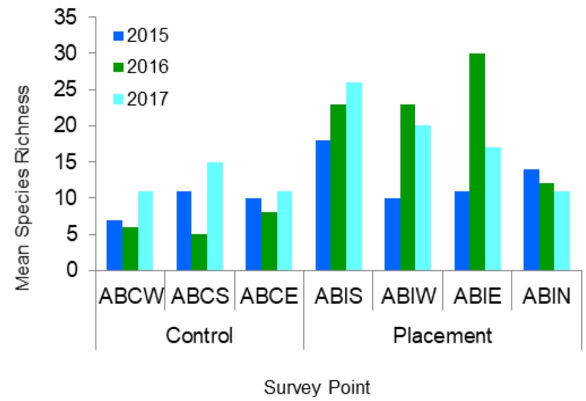


Figure 56. *Avian species richness at Avalon by survey point pre-placement (2015) and post-placement (2016 and 2017) during point-count avian surveys.*

At Fortescue, the proportion of individuals belonging to each guild shifted between years in both the control and placement areas. When interpreting the following percentages, it is important to note the total birds counted during the survey efforts, as these vary widely.

Within the control areas, between 2015, 2016, and 2017, there was a shift in the gull community (3, 32, and 10% respectively), and a sharp decline in the rail community (31, 2, and 2% respectively). Passerines declined from 2015 to 2016, then nearly doubled in 2017 (38, 22, and 53% respectively), while wading birds increased from 2015 to 2016, then decreased in 2017 (9, 21, and 4% respectively). Shorebirds increased within the control area from 2015 to 2017 (4, 13, and 16% respectively).

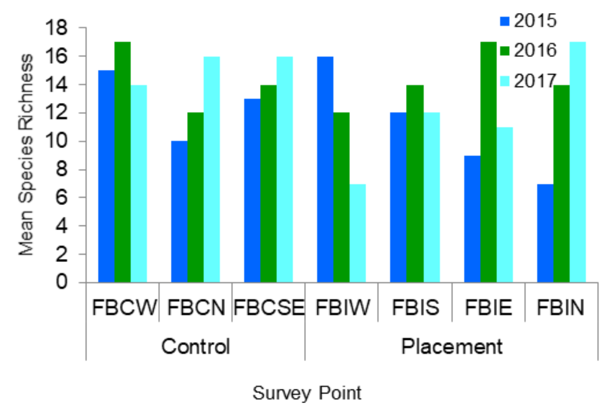


Figure 57. *Avian species richness at Fortescue by survey point pre-placement (2015) and post-placement (2016 and 2017) during point-count avian surveys.*

Within the placement areas, there were shifts in the rail, gull, and passerine guilds (2015 is baseline data, PRE placement of dredged material). Rails declined from 2015 to 2017 (25, 2, and 1%) at placement and control sites, while the proportion of gulls increased dramatically from 2015 to 2016 (5 to 55%), then decreased dramatically from 2016 to 2017 (55 to 7%). Passerines (sparrows, swallows, blackbirds, starlings, crows, wrens, grackles) declined dramatically from 2015 to 2016 from 60 to 21% then increased dramatically from 2016 to 2017 from 21 to 55%. All other guild proportions remained similar between years, though a small increase in shorebirds is noted within placement areas from 2015 to 2017 (2, 13, and 14% respectively).

Conclusions

- The placement of dredged material on the marsh surface initially made acute changes to the habitat available to birds using each project site.

- Shorebirds increased in abundance post-placement, most likely due to the creation of open, sparsely vegetated areas, which they prefer for foraging and resting.
- Marsh-dependent species, such as clapper rail (*Rallus crepitans*), willet (*Tringa semipalmata*), and seaside sparrow (*Ammodramus maritimus*), which require dense vegetation for nesting, were not recorded in large numbers in the placement areas at any of the sites post-placement, though a few pairs remained. These numbers started to rebound after two years post-placement. This could indicate that the species displaced by the placement of dredged material may return to the enhanced areas over time, as vegetation and other habitat components recover.
- Marsh-dependent species, including willet and seaside sparrows, nested in marsh vegetation at the Ring Island ENH that established on the site.
- The ENH site required vegetation management after three years from initial placement to maintain habitat for species initially targeted by the design.
- The TLP areas supported fewer species and fewer nests compared to the ENH, but the number of species nesting in these areas increased over the five-year monitoring period. Reproductive success was also lower, but the reproductive outcomes were comparable to control areas in the marsh.
- Based on these results, we can address the following monitoring questions: 1) "Does the marsh ecosystem (i.e., structure and functions) recover or show uplift within two to three years of dredged material placement? (And the related "How long does it take for the marsh to recover and achieve uplift?")" and 2) "Did differences in structural factors correlate with recover and uplift?"
    - 1) Avian populations overall demonstrated a quick response to placement treatments, with counts increasing at both Ring Island and Avalon within a short timeframe. At Ring Island, multiple species nested on the elevated nesting habitat in all years and within the thin-layer placement areas in four of the five years monitored. Reproductive success at ENH was high in all years except 2019 when predation was high, and the nesting platform became densely vegetated. Black skimmers and least terns, both endangered species in New Jersey, and a large colony of common terns nested successfully in two to three years. Avalon demonstrated significant increases in avian populations within one-year post-placement, and Fortescue populations were insignificantly different between control and placement areas in all years of monitoring.
    - 2) The habitat shifts caused by placement resulted in changes in species presence, resulting in increases in species that preferred open, bare habitat for foraging. More secretive species that prefer dense vegetation were negatively impacted at first. Elevation and corresponding vegetation cover were found to be important factors in avian population recovery and uplift.

11. Nekton Monitoring

Evan Thomas Kwityn[1,2], John A. (Jack) Szczepanski[1]
[1]Princeton Hydro, [2]Montclair State University

Salt marsh nekton communities encompass an array of species that aid and contribute to the local estuary ecosystem. Nekton are defined as organisms that are free-swimming fish, shrimp, and crabs (Raposa & Talley, 2012). Nekton take advantage of particular habitats at different life-history stages in salt marshes for foraging, reproduction, refuge from predation, and overwintering (Litvin and Weinstein, 2003). Utilizing ubiquitous quantities of habitats within a salt marsh, nekton communities are able to allocate the transition of energy to other regions of an estuary. Nekton species are ubiquitous, diversified, and abundant within salt marshes and provide essential food resources for opportunistic avian fauna and economically important fishery species (Barbier et al., 2011). Natural salt marsh degradation and human influences have impacted the life history of nekton. Upon successful tidal salt marsh restoration practices, marshes should experience improved hydrologic and biogeochemical components that would promote nekton habitat functions and aggregations associated with high species diversification found in natural salt marsh ecosystems (Raposa & Talley, 2012).

Monitoring Design

Nekton surveys were conducted during the growing seasons in 2015 (preplacement), 2016, and 2017 (both post-treatment) at Avalon and Fortescue. Sampling occurred during the annual vegetative growing season in September 2015, from August-November in 2016, and July-August 2017. These sampling events were correlated with spring tide events to ensure periods of active site use rather than diapause or reduced tidal inundation. Replicated sampling around the middle to late vegetative growing season also allowed time for the restored placement habitats to stabilize and partially revegetate after the short-term impacts of sediment placement. Each sampling event consisted of 40 or more sampling locations and took roughly five days to complete. A sampling day varied substantially given the number of sampling locations, the extent of processing, as well as the logistics (e.g., sampling at proper water depths for ebbing and flooding) of accessing and moving about the project sites. The metrics collected during sampling events are listed in Table 29.

Table 29. *Design and Metrics used for Nekton Monitoring at Avalon and Fortescue, NJ.*

Design	Metrics
B-A-C-I and Site-Specific Habitat (Pools, Marsh Shoreline, Subtidal Creek, Tidal Creek)	Species Abundance Species Density Species Richness Length
Water Quality	Temperature Salinity Dissolved Oxygen

Nekton monitoring followed a stratified random sampling design. Sampling was divided into two categories, control (i.e., controls) and enhanced (i.e., experimental) treatment sites at four habitats on a coastal salt marsh. Control locations are identified as locations unaffected by salt marsh enhancement, while enhanced sites included the beneficial placement of thin-layer dredged material to combat salt marsh subsidence and increase elevation. The site-specific habitats included: 1) marsh pools, 2) subtidal creeks and/or marsh ditches, 3) tidal creeks, and 4) the marsh shoreline edge. Five sample points in both control and enhanced categories were selected at random within each habitat. A total of 40 or more sampling locations were investigated per sampling event depending upon the location of dredged material during salt marsh enhancement. Candidate sample locations were determined before fieldwork but assessed and selected in the field. Sampling was repeated at the same location during post-placement sampling events where dredged material has not altered features that resulted in areas without surficial hydrologic activity. If a control sampling location became influenced by dredged material during restoration, it was changed and categorized as an enhanced sampling location. If restoration altered extant habitats, new sample locations were selected at random in the field within the affected habitat strata for post-placement sampling events.

Nekton were sampled using a standard 1.0 m x 1.0 m x 0.5 m throw trap with a 1.0 m x 0.5 m dip net for sampling large open bodies of water such as salt marsh pools, tidal creeks, and the marsh shoreline edge. Standardized ditch nets were also constructed and used for sampling narrow tidal channels and mosquito ditches up to 1.0 m wide with a depth of 1.0 m (Neckles et al., 2013; James-Pirri et al., 2012). Temperature, salinity, and dissolved oxygen were taken with a Hydrolab Quanta Multi-Probe Meter in concurrence with nekton sampling. It is important to indicate that water quality was only taken in marsh pools prior to the deployment of the throw trap. Water quality taken before sampling in marsh pools was done to minimize disturbance of sediment resuspension. All other water quality measurements for each habitat (i.e., tidal creeks, subtidal creeks, ditches, and the shoreline edge) were taken after deployment of the throw trap or ditch net. Before processing a throw trap, an estimation of vegetative percent cover and a direct measurement of water depth were evaluated for potential nekton habitat. Processing of throw traps consisted of sweeping a dip net through the throw trap until three consecutive dip net sweeps yielded a catch of zero nekton. Ditch traps were deployed for at least 30 minutes before retrieval and processing. Ditch nets were also measured for the dimensions of the trap to standardize nekton density calculations with throw traps.

Results

Results of these nekton surveys were published as part of a Master of Science thesis by Evan T. Kwityn,[11] and only the main results are discussed here.

In 2015, before salt marsh restoration, there were no significant differences between restored and unrestored treatment, or habitat locations for Avalon, New Jersey. Daggerblade grass shrimp (35%) were the dominant species, but other principal fish species, such as Atlantic silversides (23%), mummichogs (16%), and sheepshead minnows (16%) also demonstrated relatively high densities and abundance (Table 30).

At Fortescue, there were no significant differences between restored and unrestored treatment locations or habitats within the 2015 dataset (Table 31). Unlike Avalon, the nekton community at Fortescue was dominated by mummichogs (70%). The nekton community at Fortescue was also comprised of

[11] Assessing the Influence of Salt Marsh Enhancement on Nekton Communities (montclair.edu)

daggerblade grass shrimp (22%), Atlantic silversides (10%), and sheepshead minnows (8%). It is important to note that only one sampling event was conducted at each site (i.e., Avalon and Fortescue, NJ) and is likely not an accurate representation of the annual nekton assemblage for 2015.

Table 30. *Average nekton density (number of individuals per m^2 ± standard error) in control and placement areas at Avalon, NJ for 2015-2016.*

Species (Common Name)	Site: Avalon			
	2015		2016	
	Unrestored n=20	Restored n=20	Unrestored n=36	Restored n=54
Fundulus heteroclitus (mummichog)	9.50 ± 5.34	9.00 ± 4.59	13.61 ± 7.11	7.29 ± 3.63
Fundulus majalis (striped killifish)	5.10 ± 2.07	4.95 ± 2.3	3.71 ± 1.12	10.48 ± 6.55
Lucania prava (rainwater killifish)	0	0	5.24 ± 2.41	0.40 ± 0.21
Menidia menidia (Atlantic silverside)	16.05 ± 9.73	16.75 ± 6.99	1.78 ± 0.91	1.81 ± 0.96
Palaemonetes pugio (daggerblade grass shrimp)	6.50 ± 2.99	26.20 ± 10.88	29.00 ± 7.03	19.31 ± 6.87
Carcinus maenas (green crab)	0	0.05 ± 0.05	0	0
Crangon septemspinosa (sevenspine bay shrimp)	0	0	0.07 ± 0.05	0.08 ± 0.08
Callinectes sapidus (blue crab)	0.75 ± 0.40	0.70 ± 0.51	0.73 ± 0.18	0.44 ± 0.17
Panopeus herbstii (Atlantic mud crab)	0.30 ± 0.25	0.05 ± 0.05	0.31 ± 0.11	0.1 ± 0.05
Cyprinodon variegatus (sheepshead minnow)	12.70 ± 6.33	1.95 ± 1.33	4.61 ± 1.42	1.85 ± 1.22
Gobiosoma bosc (naked goby)	0.05 ± 0.05	0	0.27 ± 0.16	0.1 ± 0.1
Mugil cephalus (striped mullet)	0.05 ± 0.05	0	0	0.02 ± 0.02
Uca pugnax (Atlantic marsh fiddler crab)	0	0.09 ± 0.08	0	0.06 ± 0.06
Micropogonias undulatus (Atlantic croaker)	0	0	0.02 ± 0.02	0
Pogonias cromis (black drum)	0	0.08 ± 0.08	0	0
Brevootia tyrannus (Atlantic menhaden)	0	0	0.10 ± 0.10	0
Syngnathus fuscus (northern pipefish)	0	0	0.05 ± 0.05	0
Total fishes	43.50 ± 15.35	33.33 ± 9.04	29.39 ± 7.84	21.96 ± 10.45
Total decapods	7.55 ± 2.95	27.91 ± 10.85	30.12 ± 7.07	19.98 ± 6.92
Total nekton	37.85 ± 15.28	54.50 ± 14.09	59.05 ± 10.04	41.44 ± 14.07

Nekton were collected throughout using a throw trap and ditch nets
Densities are not log transformed
n= Total number of sampling locations

Table 31. *Average nekton density (number of individuals per m^2 ± standard error) in control and placement areas at Fortescue, NJ for 2016.*

Species (Common Name)	Site: Fortescue			
	2015		2016	
	Unrestored n=20	Restored n=19	Unrestored n=34	Restored n=35
Fundulus heteroclitus (mummichog)	25.68 ± 9.96	33.90 ± 12.86	3.59 ± 0.90	7.21 ± 3.67
Fundulus majalis (striped killifish)	0	0	6.71 ± 2.15	14.24 ± 7.78
Lucania prava (rainwater killifish)	0	0	2.18 ± 1.00	0.94 ± 0.54
Menidia menidia (Atlantic silverside)	3.05 ± 1.96	2.15 ± 2.05	1.03 ± 0.58	0.30 ± 0.20
Palaemonetes pugio (daggerblade grass shrimp)	6.86 ± 3.68	4.55 ± 2.33	34.50 ± 12.87	19.30 ± 9.86
Crangon septemspinosa (sevenspine bay shrimp)	0.77 ± 0.68	0	0.12 ± 0.12	0.67 ± 0.41
Callinectes sapidus (blue crab)	0.05 ± 0.05	0	0.94 ± 0.29	0.58 ± 0.19
Panopeus herbstii (Atlantic mud crab)	0.11 ± 0.08	0	0.09 ± 0.07	0.03 ± 0.03
Cyprinodon variegatus (sheepshead minnow)	3.32 ± 3.09	0.78 ± 0.59	2.62 ± 1.15	1.30 ± 0.88
Gobiosoma bosc (naked goby)	0	0	0.09 ± 0.07	0
Uca pugnax (Atlantic marsh fiddler crab)	0	0	0.09 ± 0.05	0
Micropogonias undulatus (Atlantic croaker)	0.14 ± 0.1	0.05 ± 0.05	0	0
Trinectes maculatus (hogchoker)	0	0	0.09 ± 0.07	0
Total fishes	32.18 ± 10.97	39.90 ± 13.39	16.29 ± 3.63	24.00 ± 11.61
Total decapods	7.77 ± 3.71	4.55 ± 2.33	35.74 ± 12.9	20.58 ± 9.93
Total nekton	34.09 ± 11.36	40.15 ± 14.09	52.03 ± 14.55	44.58 ± 14.84

Nekton were collected throughout using a throw trap and ditch nets
Densities are not log transformed
n= Total number of sampling locations

Table 32. *Total nekton abundance in tidal creeks, pools, subtidal creeks, and the shoreline edge at Avalon, NJ 2016.*

Species (Common Name)	2016 Avalon							
	Unrestored				Restored			
	Tidal Creek n=6	Pools n=12	Shoreline Edge n=10	Subtidal Creek n=8	Tidal Creek n=14	Pools n=18	Shoreline Edge n=10	Subtidal Creek n=12
Fundulus heteroclitus (mummichog)	22	21	499	8	122	6	4	210
Fundulus majalis (striped killifish)	32	68	32	12	135	22	2	343
Lucania parva (rainwater killifish)		214	1			14		3
Menidia menidia (Atlantic silverside)	6	66	1		57		26	5
Palaemonetes pugio (daggerblade grass shrimp)	300	179	657	52	394	5	403	112
Carcinus maenas (green crab)								
Crangon septemspinosa (sevenspine bay shrimp)	3				4			
Callinectes sapidus (blue crab)	13	10	5	2	9	8	4	
Panopeus herbstii (Atlantic mud crab)		2	7	2	1			3
Cyprinodon variegatus (sheepshead minnow)	6	179	4		21	60		8
Gobiosoma bosc (naked goby)	1		10		2		1	1
Mugil cephalus (striped mullet)	1				1			
Uca pugnax (Atlantic marsh fiddler crab)								3
Micropogonias undulatus (Atlantic Croaker)	1							
Brevootia tyrannus (Atlantic menhaden)	4							
Syngnathus fuscus (northern pipefish)			2					
Pogonias cromis (black drum)								

n= number of sampling sites within each habitat

In 2016 following the application of thin-layer dredged material, an additional sampling event was conducted at both Avalon and Fortescue, NJ. This additional sampling event made a total of two sampling events per site per year. Sampling at Avalon, NJ was held during the middle and end of the annual growing season, during August and October of 2016.

At Avalon, there were significant differences between datasets (e.g., restored vs. unrestored) for percent cover for nekton density (Table 32). The most significant habitat among restored and unrestored treatments observed at Avalon was found between marsh pools. Unrestored salt marsh pools contained 739 nekton individuals and restored salt marsh pools contained 115 nekton individuals. Among restored marsh pools sheepshead minnows occupied the highest total abundance of 60 individuals. Daggerblade grass shrimp (47%) were observed to be the most dominant nekton and decapod species. Mummichogs (20%), striped killifish (15%), and sheepshead minnows (6%) encompassed the majority of the nekton community at Avalon.

In 2016, Fortescue displayed significant nekton density among all habitats (e.g., marsh pools, ditches, tidal creeks, and the marsh shoreline) (Table 33). However, no significant trends in nekton density were observed between restored and unrestored salt marsh restoration treatments. The most dominant nekton species at Fortescue was daggerblade grass shrimp (56%). Other common nekton species observed at Fortescue included striped killifish (22%), and mummichogs (11%).

In 2017 at Avalon there was a total of 12 (i.e., 8 fish and 5 decapod species) nekton species observed (Table 34). Fish species consisted of 63% and decapods consisted of 37% of the nekton community. The most dominant species was the daggerblade grass shrimp (34%). Other common nekton species observed at Avalon included striped killifish (28%), sheepshead minnow (13%), and mummichogs (10%).

At Fortescue in 2017, there was a total of 10 (i.e., 6 fish and 4 decapod species) nekton species observed (Table 35). Fish species consisted of 41% and decapods consisted of 59% of the nekton community. The most dominant species was the daggerblade grass shrimp (58%). Other common nekton species observed at Fortescue included striped killifish (19%), mummichogs (9%), Atlantic silverside (6%), and sheepshead minnow (5%).

Additional data and figures can be found in Appendix B.

Table 33. *Total nekton abundance in tidal creeks, pools, subtidal creeks, and the shoreline edge at Fortescue, NJ 2016.*

Species (Common Name)	2016 Fortescue Control Tidal Creek n=5	Control Pools n=10	Control Shoreline Edge n=9	Control Ditch n=10	Treatment Tidal Creek n=5	Treatment Pools n=10	Treatment Shoreline Edge n=10	Treatment Ditch n=8
Fundulus heteroclitus (mummichog)	41	39		42	168	36	1	33
Fundulus majalis (striped killifish)	51	108	2	67	283	88	4	95
Lucania prava (rainwater killifish)		73		1		18		13
Menidia menidia (Atlantic silverside)	2	13	20		9		1	
Palaemonetes pugio (daggerblade grass shrimp)	384	671	63	55	221	102	310	4
Carcinus maenas (green crab)								
Crangon septemspinosa (sevenspine bay shrimp)			4			10	12	
Callinectes sapidus (blue crab)	2	11	15	4	5	4	10	
Panopeus herbstii (Atlantic mud crab)	1			2				1
Cyprinodon variegatus (sheepshead minnow)		80		9		31		12
Gobiosoma bosc (naked goby)			3					
Mugil curema (white mullet) or striped								
Uca pugnax (Atlantic marsh fiddler crab)	1			2				
Micropogonias undulatus (Atlantic Croaker)								
Brevootia tyrannus (Atlantic menhaden)								
Syngnathus fuscus (northern pipefish)								
Trinectes maculatus (hogchoker)			3					
Pogonias cromis (black drum)								

n= number of sampling sites within each habitat

Table 34. *Average nekton density (number of individuals per m² ± standard error) in tidal creeks, pools, and the marsh shoreline edge at Avalon, NJ for 2017.*

Species (Common Name)	*Site Avalon 2017 Control n=41*	*Enhanced n=54*
Fundulus heteroclitus (mummichog)	15.74 ± 10.37	2.26 ± 0.86
Fundulus majalis (striped killifish)	22.82 ± 14.80	14.56 ± 10.91
Lucania prava (rainwater killifish)	4.82 ± 2.13	0.20 ± 0.13
Menidia menidia (Atlantic silverside)	7.44 ± 2.47	1.31 ± 0.58
Palaemonetes pugio (daggerblade grass shrimp)	30.10 ± 12.10	9.44 ± 5.84
Carcinus maenas (green crab)	0	0
Crangon septemspinosa (sevenspine bay shrimp)	0	0.09 ± 0.06
Callinectes sapidus (blue crab)	2.95 ± 0.81	1.01 ± 0.30
Panopeus herbstii (Atlantic mud crab)	0	0
Cyprinodon variegatus (sheepshead minnow)	18.32 ± 8.78	1.20 ± 0.53
Gobiosoma bosc (naked goby)	0.41 ± 0.19	0.13 ± 0.08
Mugil cephalus (striped mullet)	0	0
Uca pugnax (Atlantic marsh fiddler crab)	0.16 ± 0.08	0.09 ± 0.05
Micropogonias undulatus (Atlantic croaker)	0	0
Pogonias cromis (black drum)	0	0
Brevootia tyrannus (Atlantic menhaden)	0.15 ± 0.09	0
Syngnathus fuscus (northern pipefish)	0.05 ± 0.05	0
Eurypanopeus depressus (flatback mud crab)	0.08 ± 0.08	0
Total fishes	32.69 ± 8.80	23.72 ± 11.86
Total decapods	30.47 ± 7.61	20.94 ± 6.46
Total nekton	62.31 ± 11.00	40.59 ± 14.93

Nekton were collected throughout using a throw trap and ditch nets
Densities are not log transformed
n= Total number of sampling locations

Table 35. *Average nekton density (number of individuals per m^2 ± standard error) in tidal creeks, pools, ditches, and the marsh shoreline edge at Fortescue, NJ for 2017.*

Species (Common Name)	Site Fortescue 2017	
	Control n=41	Enhanced n=47
Fundulus heteroclitus (mummichog)	5.46 ± 2.90	2.41 ± 0.76
Fundulus majalis (striped killifish)	9.18 ± 4.08	6.99 ± 2.69
Lucania prava (rainwater killifish)	0.61 ± 0.21	0.46 ± 0.21
Menidia menidia (Atlantic silverside)	1.39 ± 0.50	2.67 ± 1.92
Palaemonetes pugio (daggerblade grass shrimp)	25.44 ± 9.89	18.43 ± 9.05
Crangon septemspinosa (sevenspine bay shrimp)	0	0
Callinectes sapidus (blue crab)	0.59 ± 0.18	0.12 ± 0.10
Panopeus herbstii (Atlantic mud crab)	0	0.02 ± 0.02
Cyprinodon variegatus (sheepshead minnow)	3.29 ± 1.54	0.88 ± 0.68
Gobiosoma bosc (naked goby)	0.03 ± 0.03	0
Uca pugnax (Atlantic marsh fiddler crab)	0.18 ± 0.09	0
Micropogonias undulatus (Atlantic croaker)	0	0
Trinectes maculatus (hogchoker)	0	0
Total fishes	19.95 ± 7.12	13.43 ± 4.07
Total decapods	26.21 ± 9.90	18.57 ± 9.14
Total nekton	44.18 ± 12.13	30.84 ± 9.81

Nekton were collected throughout using a throw trap and ditch nets
Densities are not log transformed
n= Total number of sampling locations

Conclusions

- In 2016, total fish, total decapods, and total nekton populations demonstrated a significant downturn in densities following placement of dredged material in both Avalon and Fortescue. Average fish density remained higher in control areas than in placement areas in 2017. This was true for all fish species, except Atlantic silversides. However, the density of total fish, decapods, and nekton was not significantly different in Fortescue across control and placement areas in 2017. We found the populations of total fish and total decapods to be proportionate across the control and placement areas. Success of these populations could be due to frequent tidal inundation at high tide, increasing the mobility of fish.
- Striped killifish and mummichogs were the most dominant fish species in 2016 and 2017 at both Avalon and Fortescue. In Avalon, although striped killifish were most dense within placement habitats in 2016, both striped killifish and mummichog populations had greater densities in control areas over placement areas, especially within tidal creeks, subtidal creeks, and the shoreline edge by 2017. However, both fundulid species had the greatest density within placement areas tidal creeks, while other habitat types within the placement areas had almost no individuals in 2017. We determined that these population preferences are likely due to differences in vegetative cover. Mummichog density was significantly correlated with a higher cover of *Spartina alterniflora*, *Ulva lactuca*, *Gracilaria* spp., and *Ruppia maritima*. Striped killifish were found to have a preference for sandier substrates with less vegetation, likely as a result of their reproductive requirements to lay eggs within sandy substrates within intertidal pools.
- At Fortescue, species richness of the whole nekton community in 2016 and 2017 did not significantly change. However, overall, there was a greater number of nekton species found in control areas than in placement areas. Some species were only found in control areas.

- At Avalon in 2017, nekton density was variable among the pools, possibly due to differences in sediment substrate, depth, tidal restrictions, or seasonal behavior of the fish.
- Daggerblade grass shrimp had a high abundance in placement tidal creeks compared to control tidal creeks in 2016. This is possibly due to the higher nutrient levels found within placement area tidal creeks. Sediments with elevated nutrient levels could result in higher abundances of macroinvertebrates, allowing fish predation to also increase. This would allow for higher trophic-level fish species to be successful within these habitats.
- Based on these results, we can address the following monitoring question: “Does the marsh ecosystem (i.e., structure and functions) recover or show uplift within two to three years of dredged material placement? (And the related “How long does it take for the marsh to recover and achieve uplift?”)”
    - There was a shift in nekton species after the placement of sediment. Some species were lost from placement areas and are currently only found in control areas. Habitat type and corresponding sediment and nutrient conditions seemed to be significant drivers of species presence and population counts.

Additional Monitoring

Soil Sampling and Well Installation

Authors: U.S. Army Engineer Research and Development Center

Additional monitoring that was not part of the Monitoring Plan was conducted by the U.S. Army Engineer Research and Development Center (ERDC). This monitoring focused on the physical, nutrient, and biogeochemical processes of the soil. Prior to dredged material placement, core samples were collected in vegetated and open water areas within the dredged material placement areas and adjacent control regions of the marsh. The monitoring design provides the data needed to investigate baseline sediment property differences between vegetated and open water features in the marsh, as well as detect changes within control and dredged material placement areas.

Monitoring Design

Five locations were sampled by ERDC for the metrics in Table 36 across the marsh at the Avalon site only. Two locations were in areas that received dredged material (i.e., placement), and three were in areas that did not receive dredged material (*i.e.*, control). Within each sampling location, three vegetated and three pool locations were identified using a random sampling design in ArcGIS before the first field sampling. The same locations were monitored in subsequent years.

Water level data loggers were installed in conjunction with the three vegetated and three pool soil sampling locations in one placement area and one control area for a total of 12 water level data loggers. Two atmospheric loggers were also installed. Water level data is recorded every 15 minutes for approximately one month.

Table 36. *Design and metrics used by ERDC for soil and well monitoring at the Avalon project site.*

Design	Sediment Metrics	Nutrient Metrics	Microbial Metrics
Post-Construction	Depth of Dredged Material	Organic Matter	Biomass Carbon
Control-Impact	Bulk Density Root Distribution	Total Phosphorous Extractable Nitrogen	Biomass Nitrogen Potentially Mineralizable Nitrogen
Once Annually During Peak Growing Season (June to September)	Grain Size Moisture Content	Total Dissolved Nitrogen Dissolved Organic Carbon Total Nitrogen and Carbon Extractable Ammonium Soluble Reactive Phosphorus	

The results from this monitoring were published by ERDC and are not discussed here.[12]

[12] https://doi.org/10.1016/j.ecoleng.2018.05.012

Data Evaluation - Adaptive Management Monitoring

During the monthly site inspection in July 2016, several new areas of vegetation die-off were observed at the Avalon project site. The vegetation in the die-off areas had survived the initial placement of sediment. Most of these areas were along the outside edge of the perimeter containment, covered with a thin layer of dredged material that had passed through the containment, and surrounded by healthy marsh vegetation. The immediate cause of the vegetation die-off was not apparent, though it was suspected that the dredged material or containment had altered the hydrology and/or the chemistry of the marsh soil, surface water, and/or groundwater to the degree that vegetation could not survive. Two adaptive management monitoring plans were developed to further investigate the die-off areas:

1) The *Sediment Characterization and Analytical Testing* plan was developed to generally characterize the placed dredged material and help determine which factors might contribute to vegetation success and die-off.
2) The *Surface and Groundwater Chemistry* plan was developed to determine if the dewatering and subsequent consolidation of dredged material had created ponded water and areas of altered water chemistry that may be stressful for vegetation.

Prior to initiating these monitoring plans, a soil scientist with the Natural Resources Conservation Service (NRCS) joined the Project Team on-site and conducted a field study to observe and examine the placed dredged material. This NRCS study looked at soil horizon depths, horizonation and horizon boundaries, soil texture, fluidity, Munsell Soil color, hydrogen sulfide odor, pH, rock and shell fragment percentages, peroxide color change, and organic matter content.

This adaptive management monitoring at the Avalon project site was conducted to further identify any major variances in soil physical and chemical properties between sampling points at different stages of vegetation recovery and during periods of wet and dry weather. It was believed by the Project Team that this information was important to more accurately assess the potential causes of vegetation die-off and to better understand why some areas revegetated more quickly than others.

12.1 Sediment Characterization and Analytical Testing

Author: Princeton Hydro

Methods

Sediment (*i.e.*, placed dredged material) at Avalon was studied by Princeton Hydro using the metrics in Table 37 Samples of dredged material (surface top 6 inches) were collected to evaluate the potential importance of the following parameters on vegetation survival and growth: (1) placement elevations, (2) dredged material physical characteristics, and (3) dredged material chemical characteristics (metals, nutrients). 31 samples were collected within containment in dredged material placement areas, six in vegetation die-off areas, and three VEG samples in placement areas with 100% vegetative cover. The sample data were also compared to the 2015 pre-placement marsh surface soil sample data, though the number of samples collected (n), the collection method, and the locations of the samples differed.

Table 37. *Design and metrics used for sediment characterization and analytical testing at the Avalon project site.*

Design	Metrics
Post-Construction Adaptive Management Monitoring	Sulfates Nitrate Total Phosphorus Target Analyte List (TAL) of 23 Metals Particle Size Organic Content pH Unit Weight Porosity

Results

The only consistent difference in the post-construction (2017) marsh surface sediment physical parameters at Dredged Material Placement Areas A and D within containment and in the vegetation die-off areas (outside containment) was the significantly greater mean % sand composition within containment (51.4% and 44.1%) compared to that in the die-off areas (7.3% and 17.4%).

In general, the mean concentrations of metals were similar in the 2015 marsh, 2017 marsh within containment, and 2017 vegetation die-off area (outside containment) surface sediment samples.

However, the mean concentrations of arsenic ($p < 0.05$), chromium ($p < 0.01$), silver ($p < 0.01$), and zinc ($p < 0.01$) in the 2017 vegetation die-off marsh surface sediment samples were significantly greater (t-tests) than the mean concentrations of the marsh 2015 surface sediment samples. In addition, the mean % sand composition of the surface sediment from the vegetation die-off areas (14.0%) was significantly less (t-test; $p < 0.0001$) than that of the 2015 marsh surface sediment samples (45.2%). The mean % sand composition of the sediment from the vegetation die-off areas was less than that of the 2017 within containment sediment samples (35.3%), but not significantly different (t-test; $p > 0.05$).

Each of the three VEG samples was paired with a nearby sample within containment in which vegetation had not recovered. The values of the sediment physical parameters (% sand, % moisture, bulk density, porosity, and elevation), TAL metals, and nutrients (total phosphorus, nitrate, and sulfates) were not significantly different between the pairs of samples. Similarly, the concentrations of total phosphorus, nitrate, and sulfate did not statistically differ between the VEG samples and the die-off samples.

In general, the mean concentrations of the TAL metals, nutrients, and sediment physical parameters in the VEG surface sediment samples were similar to those in the sediment samples from the vegetation die-off areas at Area A and Area D. However, beryllium ($p < 0.05$) and zinc ($p < 0.05$) were significantly higher, and vanadium lower ($p < 0.05$; t-tests) in the vegetation die-off samples.

While a correlation between metal or nutrient concentration and vegetative cover was not established by this study, elevation exhibited the strongest correlation with plant cover (percent) within the containment area. Specifically, vegetation recruitment and cover at sample locations collected between elevations of 2.47 feet to 2.60 feet (NAVD88) were encouraging (Figure 58).

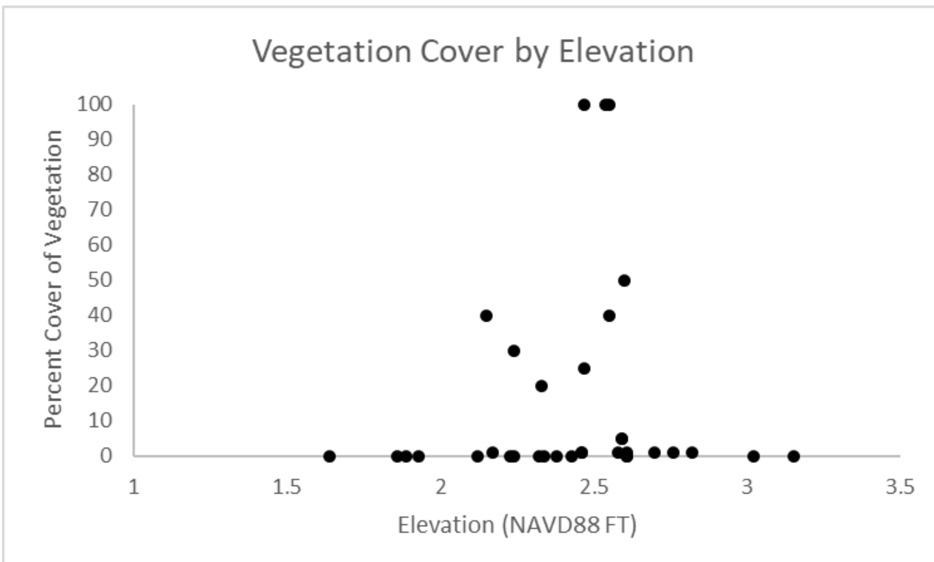


Figure 58. Elevation and Vegetation Cover at Avalon based on Princeton Hydro's characterization during the 2017 soil monitoring.

Conclusions

- Neither elevated metal, nutrient concentrations, nor the measured sediment physical parameters were found to be correlated with vegetative cover within containment. However, the number of "vegetated" samples was limited (n = 3).
- Likewise, neither elevated metal nor nutrient concentrations were found to be correlated with the incidence of vegetation die-off areas outside containment. However, the % sand composition within containment was significantly greater than that in the vegetation die-off areas.
- Vegetative recovery appears to be clustered in a specific elevation range. The effect of hydrology on vegetative regrowth at the site was not an initial study objective. However, based on this study and other site observations, it is the Team's hypothesis that vegetative regrowth is not a function of soil characteristics, but rather site drainage/hydrology.

12.2 Surface and Ground Water Chemistry

Author: Lenore Tedesco[1]
[1]The Wetlands Institute

Methods

Table 38. *Design and metrics used for surface and groundwater chemistry monitoring at the Avalon project site in 2017.*

Design	Metrics	
Control	Temperature (°C)	Dissolved Oxygen (% saturation and mg/L)
Reference	Conductivity (µs/cm)	pH
Placement	Specific Conductivity (µs/cm)	Oxidation Reduction Potential (mV)
	Total Dissolved Solids (mg/L)	Sampling Depth (m)
	Salinity (ppt)	

Water chemistry was studied in the groundwater and surface water pannes and pools at Avalon by The Wetlands Institute from 2017-2020 with metrics outlined in Table 38. Surface water and groundwater chemistry in salt marsh pannes and pools and shallow groundwater wells and piezometers were measured following the placement of dredged material at the Avalon, NJ project site every quarter to document site conditions. This monitoring program was initiated in 2017 in response to observations of vegetation die-off and surface water chemical conditions that were stressful to vegetation. There was a pause in monitoring from Fall 2017 until late Spring 2018. The longer-term monitoring then utilized a sampling strategy focused on documenting the range of water chemistry that occurs seasonally to understand if the water chemistry is controlling vegetation recovery and/or survival of vegetation plantings at the Avalon site. The water chemistry of groundwater, accessed through USACE piezometers and groundwater wells, was included in the monitoring program following initial surface water

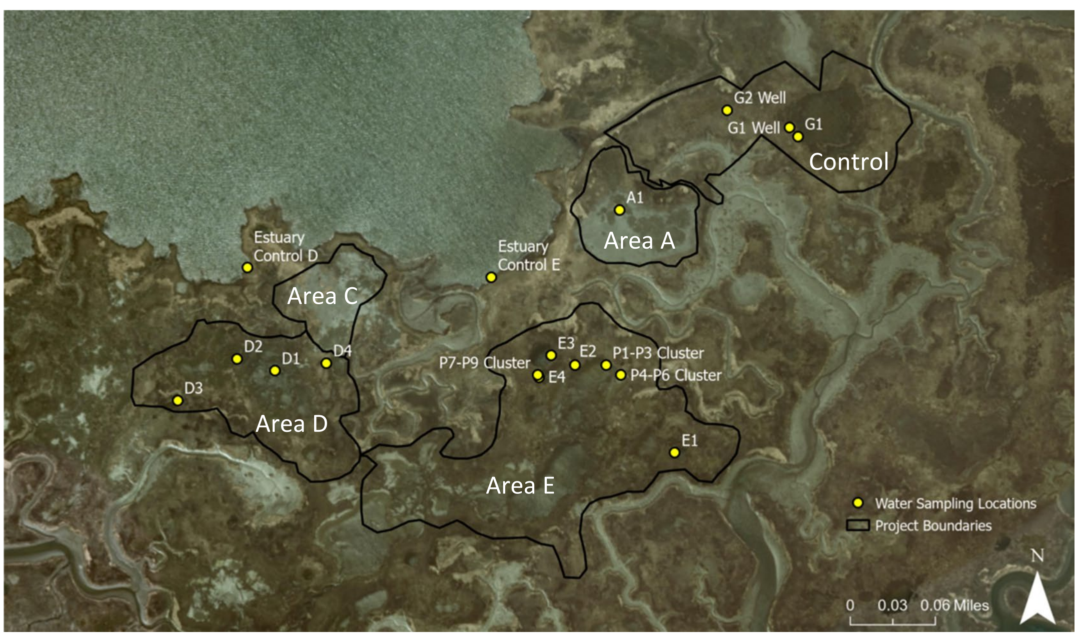


Figure 59. *Map of sampling locations for groundwater and surface water.*

monitoring. The groundwater monitoring effort was implemented in response to very high salinities measured on-site in the early spring of 2017, as well as the observation of new vegetation die-off areas. For comparative purposes, this monitoring program utilized groundwater wells and long-term monitoring data from pools and marsh depressions at a nearby site, control areas at Avalon, as well as two nearby estuary control sites (Figure 59).

Results

Surface Water Control and Reference Stations

One estuary control site was monitored at the Avalon site and one estuary control site was monitored at The Wetlands Institute (TWI). Salinity ranged from 16.8 to 41.6 parts per thousand (ppt) at the Avalon site and 19.3 to 32.6 ppt at TWI (Table 39). The lowest salinity measured at both sites followed 5-day prior rainfall events of 1.96” and 1.85”. The higher salinity in the estuary at Avalon is interesting as it is higher than salinities measured in control and reference pannes, but similar to the reference depression at TWI. The control panne at Avalon (G1) ranged from 15.6 to 37.2 ppt. The reference panne at TWI (D) ranged from 16.8 to 39.1 ppt. The reference depression at TWI (E) displayed a similarly wide range of salinities as the Avalon estuary control site (16.1 to 41.3 ppt). The mean pH for the two estuary control sites was slightly lower than the Avalon control panne (mean 7.96; 7.91; 8.14, respectively), while the reference panne and reference depression had higher mean pH (8.33; 8.35, respectively; Table 39).

Table 39. *Average, Minimum, and Maximum values for select water chemistry sampling at Avalon and The Wetlands Institute in 2017. Ground water was sampled in wells and piezometers. Surface water was measured in pools, pannes, and other depressions that apply.*

		Temperature (^oC)			Salinity (ppt)			Dissolved Oxygen (percent Sat)			pH		
		Average	Min	Max	Average	Min	Max	Average	Min	Max	Average	Min	Max
Ground Water	Placement Areas	24.5	21.7	28.0	38.2	31.8	48.5	9.4	3.1	39.5	7.0	6.3	8.1
	Control or Reference	22.9	18.1	27.3	36.5	31.7	44.3	10.6	4.1	28.1	6.8	6.5	7.0
Surface Water	Placement Areas	25.6	10.1	34.9	37.1	30.5	58.5	164.5	79.4	248.0	8.8	8.2	9.6
	Control or Reference	28.7	19.4	36.9	33.6	30.1	43.6	162.9	54.1	300.1	8.3	7.6	8.9
	Estuary Control	22.3	9.6	28.4	30.8	27.0	32.8	100.7	76.2	118.7	7.9	7.6	8.2

Dissolved oxygen saturation was highly variable among the control and reference sites (Figure 60). DO for the estuary sites were very similar, however, the control panne at Avalon (G1) had the lowest DO measurements (range 46-181%) and the reference panne and reference depressions at TWI had the highest DO measurements (295% and 369%, respectively). The reference depression at TWI was created by compaction of the marsh surface during construction and has restricted tidal flushing. The control panne at Avalon is well connected to tidal flushing and this may be moderating measured values overall. All surface control and reference sites had times of filamentous green algae to varying degrees and these occurrences affect DO and pH values in addition to the sampling timing relative to flushing conditions.

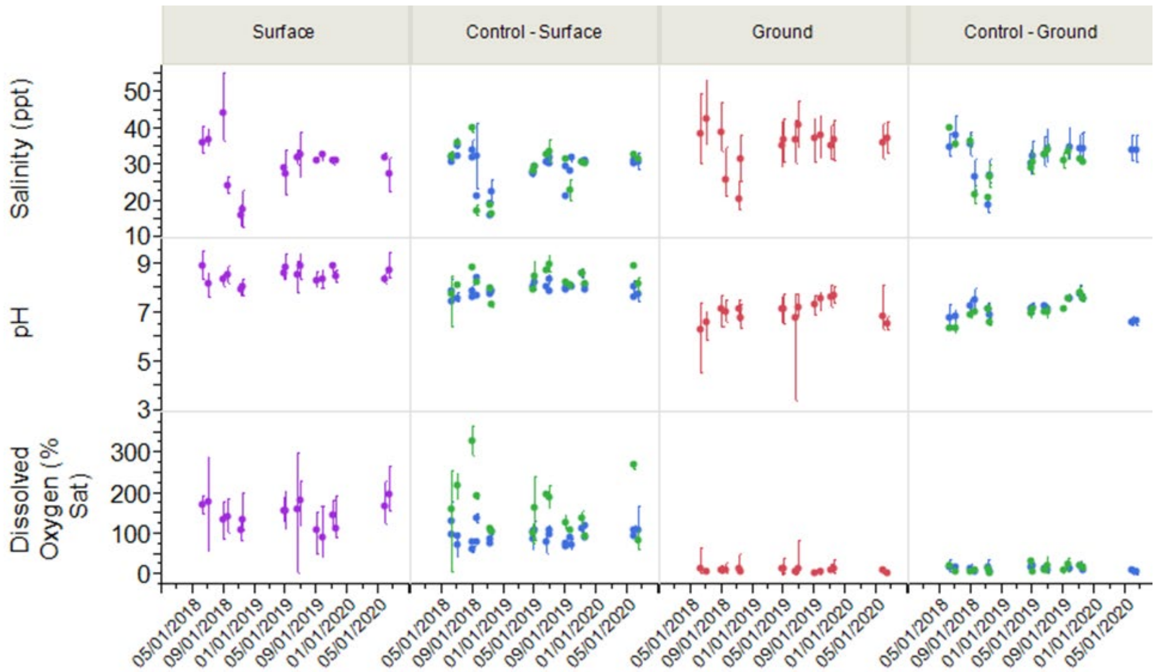


Figure 60. *Graphs of mean and range of measurements for Salinity (ppt), pH, and Dissolved Oxygen (%) of Surface Water Sites, Surface Control (Blue) and Reference (Green) Sites, Groundwater Sites and Groundwater Control (Blue) and Reference (Green) Sites, 2018-2020.*

Groundwater from Control and Reference Wells

One of the control wells (G2) located on the marsh platform at Avalon had higher maximum salinity (43.6 ppt) than all other control or reference wells (40.2 ppt TWI), the estuary control sites, and the control panne (41.6, 32.6, and 37.3 ppt, respectively; Table 34). This was measured in June of 2018 which corresponds to a time of low 5-day prior precipitation and also the time of the highest overall salinities measured during the monitoring period. The wells and control panne had the same minimum salinity (16.8 ppt). The reference wells in the marsh plain at TWI (Wells 1 and 2) showed similar salinity as the Avalon control wells (mean 31.0 vs 32.3 ppt, respectively). TWI reference wells and Avalon control wells had similar mean pH (7.1 and 7.2, respectively) and only slightly different max and min pH (Table 40).

Table 40. *Temperature (T°C), pH, Salinity (ppt) and Dissolved Oxygen (%) of Surface Water Sites, Surface Control and Reference Sites, Groundwater Sites and Groundwater Control and Reference Sites (2018-*

Site		N	Mean T (°C)	Max T (°C)	Min T (°C)	Mean pH	Max pH	Min pH	Mean Salinity (ppt)	Max Salinity (ppt)	Min Salinity (ppt)	Mean DO (% Sat)	Max DO (% Sat)	Min DO (% Sat)
Control Sites														
Avalon	Estuary Control	16	19.06	28.09	5.55	7.96	8.42	7.48	29.91	41.62	16.84	95.53	127.19	70.27
TWI	Estuary Control	16	19.85	29.66	9.18	7.91	8.65	7.44	28.48	32.60	19.34	97.13	132.63	62.57
Avalon	Control Panne	15	19.07	27.50	1.12	8.14	8.70	7.53	29.66	37.29	15.56	101.50	181.17	46.37
Avalon	Control Wells	45	17.82	25.74	8.55	7.17	8.15	6.48	32.25	43.61	16.94	14.94	38.10	0.10
Reference Sites														
TWI	Reference Panne	16	21.65	34.24	6.41	8.33	9.32	7.24	29.07	39.07	16.75	158.17	294.90	77.90
TWI	Reference Depression	16	23.66	36.68	6.89	8.35	9.07	7.41	29.29	41.31	16.06	178.67	368.80	62.97
TWI	Reference Wells	28	19.14	25.65	12.53	7.08	8.06	6.21	30.99	40.16	19.28	15.49	43.20	3.10
Beneficial Use Sites														
Avalon	Surface Water	126	21.52	37.22	3.09	8.49	9.53	7.64	29.78	55.62	12.82	147.13	300.93	5.53
Avalon	Groundwater	136	18.91	27.05	6.44	7.10	8.14	3.44	35.64	53.29	17.61	11.31	85.00	0.25

Surface Sampling Stations within Placement Areas

Salinity, pH, Temperature, and Dissolved Oxygen

Surface water site conditions varied widely at the Avalon dredge material placement site. A comparison of surface water salinity for site samples as compared with controls (excluding estuary controls) indicates that samples from sites within the dredge material emplacement areas have higher salinity and higher pH than the control and reference sites, especially early in the monitoring period. Salinity ranged from a minimum of 12.8 to a high of 58.5. The range of surface water salinity showed higher variability than the control and reference stations (Figure 60) and is especially pronounced in 2017 and 2018 and corresponded with times of stagnation both before and following containment removal. Salinity minima were associated with rainfall events and are more pronounced during neap tide conditions and are likely indicative of isolation. Salinity minima occurred at all surface water sites during the November 2018 sampling during both spring and neap monitoring visits (spaced one week apart) and were associated with 5-day prior rainfall of 1.88” during the neap monitoring and 1.96” during the spring conditions monitoring. There were no other times during the monitoring period with such persistent high rainfall amounts and salinity in 2018 was significantly lower than in all other years of the study ($p < 0.0001$ 2018, 2019; 0.033 in 2020). These results indicate that isolated surface water on the site is susceptible to rainfall events that can drive salinity variations. These rainfall events are responsible for salinity minima across the range of sample locations including the control and reference sites, but effects are most extreme at the surface sites within the placement areas. The salinity maximum was recorded at site D1 (58.5) in March 2017 and site A1 (55.6) in August 2018 with other surface sites recording their highest salinities in this timeframe, but not as extreme as these two sites. Both of these sites are isolated shallow pools with limited tidal flushing that were sampled when containment was still on sites or shortly after removal. Monitoring later during the study appears to indicate a reduction in extreme water chemistry documented as the site matures. The cause of this may be related to increased connectivity of sites with tidal channels, and/or overall maturation of the dredged material with dewatering and tidal flushing or both.

An annual effect was detected in separate tests of surface waters for pH ($F_{3,248} = 7.6$, $p < 0.001$), salinity ($F_{3,259} = 13.8$, $p < 0.001$), DO ($F_{3,259} = 3.9$, $p < 0.05$), and temperature ($F_{3,259}=11.1$, $p < 0.05$). Salinity in 2017 (35.23 ± 0.86 ppt) was significantly higher than in other years (2018: 28.09 ± 0.70 ppt, $p < 0.0001$; 2019: 30.61 ± 0.0.63 ppt; 2020: 30.61 ± 1.30 ppt, $p < 0.05$), and salinity in 2019 was higher than 2018 ($p < 0.05$).

pH ranged from a high of 9.6 (D2) with several measured pH values over 9 (at sites D1, D2, and A1) to a low of 7.6 (A1). All of these sites have demonstrated the most isolation, especially in 2017 and 2018. Overall pH values varied over slightly higher ranges than the surface control and reference sites (Figure 60). pH in surface water was significantly higher at placement sites (8.56 ± 0.04) than at control sites (8.30 ± 0.06; $p < 0.01$). Significantly lower pH was detected in 2018 (8.19 ± 0.05) compared to 2017 (8.49 ± 0.07, $p < 0.01$) and 2019 (8.47 ± 0.04, $p < 0.0001$).

The highest temperatures recorded occurred at sites D2 and D3 (37.2°C). Differences in temperature between years were significantly influenced by site type (control vs placement {$F_3 = 4.39$, $p < 0.01$}). Surface water temperatures in 2018 (22.59 ± 0.83 °C) were significantly lower than in 2017 (26.46 ± 1.01 °C, $p < 0.05$) and 2020 (27.85 ± 1.48 °C, $p < 0.05$); similarly, surface water temperatures in 2019 (19.41 ± 0.75 °C) were also significantly lower than in 2017 ($p < 0.0001$) and 2020 ($p < 0.0001$).

Dissolved oxygen values were widely variable over the sampling period (5.5 to 300.9%) with the maximum saturation reported at E4 (300.9% saturation). These findings of more extreme conditions in the surface pools are consistent throughout the study. Dissolved oxygen (% sat) was significantly affected by the interaction of water type (surface vs. groundwater) and site type (control vs placement {$F_1 = 13.64$, $p < 0.001$}). Dissolved oxygen in surface water at placement sites (150.8 ± 3.1 %) was significantly higher than at control sites (127.2 ± 3.9 %, $p < 0.001$). Where variables are moderated by flushing, variability decreased over time. For those variables that may be locally driven by biological factors (pH, DO) there does not appear to be any moderation.

When salinities were relatively low (<35 ppt), there does not appear to be a relationship between pH and salinity, and reference and control site values overlap with dredge material sites (Figure 61A). The lowest site salinities occur in 2018 along with the highest site salinities. Site salinities above 35 ppt are limited to 2017 and 2018, while site salinities are below 35 ppt throughout 2019 and 2020. High site salinities in 2017 and 2018 are also associated with pH values above 8.5. These observations support a relationship to isolation. During the monitoring period, pH in surface waters was always alkaline. No instances of acidic surface waters were documented during the study period.

The relationship between temperature and salinity of surface sites was investigated (Figure 61B) and indicates that temperature is not necessarily a driver of salinity. The fact that there is not a clear relationship between temperature and salinity also indicates that factors other than evaporation are in play and may include dredge material dewatering especially earlier in the study when dredge materials were newly emplaced. The highest salinities occur in 2017 (Figure 61B).

An evaluation of temperature and pH shows that high pH occurred more frequently in site surface water samples while lower pH (<7.75) was largely confined to reference and control stations. There are instances

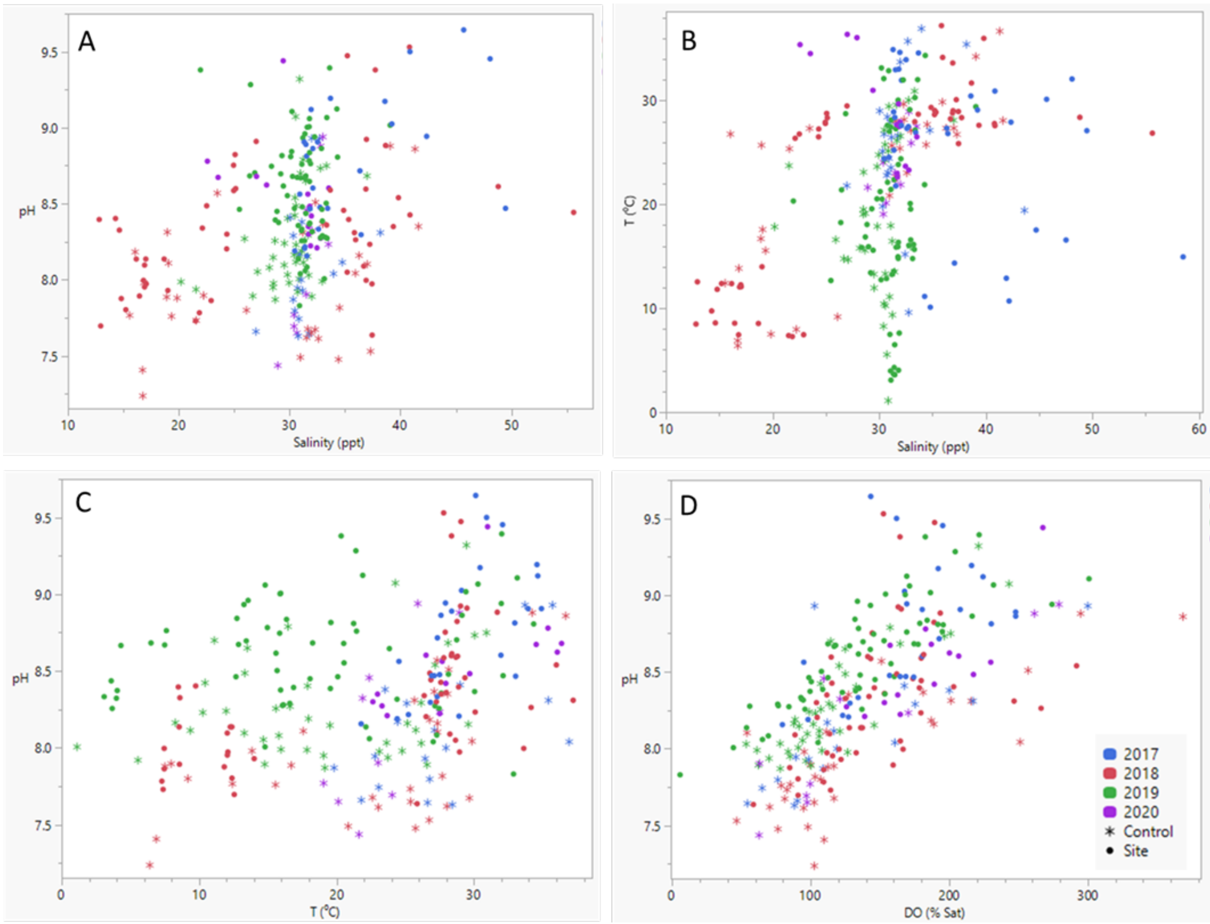


__Figure 61.__ Relationship of Site and Control Stations by Year comparing A) relative pH vs Salinity; B) Temperature vs Salinity; C) pH vs Temperature; and D) pH vs Dissolved Oxygen (%).

of high pH occurring in control and reference sites, suggesting that, at times, this is a natural phenomenon. High pH (>8.5) occurs over a wide range of temperatures (Figure 61C) and temperatures exceeded 30^{o}C in some pools at both the site and in controls during the summer attesting to the extreme conditions that occur in shallow, isolated pools regardless of their origin. These observations, along with those of the salinity relationships, indicate other factors are at work and high pH and salinity are not solely driven by temperature or evaporation.

Dissolved oxygen was commonly supersaturated and, at times, was measured as high as 300% saturation. Analysis of pH and dissolved oxygen trends show that pH increased with increasing dissolved oxygen at both the control and reference sites and the dredge material sites. Dissolved oxygen can diffuse into the water from the atmosphere and can be produced *in situ* as a chemical byproduct of photosynthesis. As photosynthesis releases oxygen, it also draws down CO_2. Carbonic acid is removed from waters and pH increases. Biologically produced alkaline pH was occurring in the isolated pools at the dredge site and in the reference depression based on direct observation of cyanobacterial mats and bubbles (Figure 62), especially during the spring, summer, and fall seasons. The distribution and extent of cyanobacterial mats varied from site to site and throughout the year. They appeared to only be limited where there was

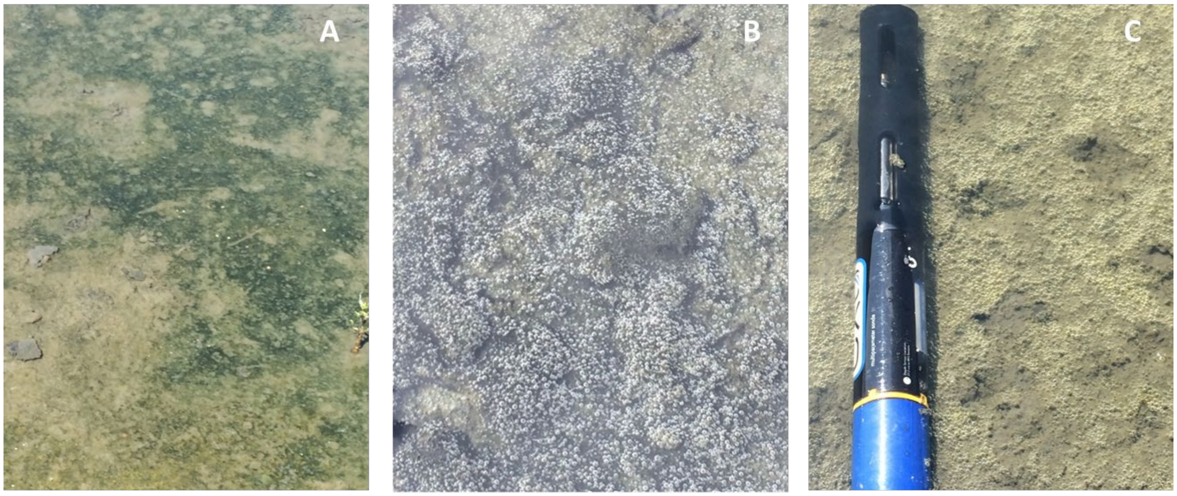


Figure 62. *Photographs of Cyanobacterial Mats and Oxygen Production. A) Cyanobacterial mat, site D1; 8/22/2017; B) Oxygen bubbles in cyanobacterial mat, well cluster P1-P3; 8/22/2017; C) Oxygen bubbles in cyanobacterial mat, site D3; 8/31/2017.*

prolonged drying or where waters were flushed enough to support grazing infauna. Cyanobacterial mats occurred naturally in pools and pannes in wetland settings and were not necessarily related to dredge material placement. While they were naturally occurring, the high pH and supersaturated oxygen conditions cyanobacterial mats caused extreme environments in isolated, shallow pools that formed in depressions in the dredge material surface. When combined with very warm water conditions (>35°C) and very high salinities (45 ppt and higher), isolated pools can be very inhospitable environments for plants and infauna. These conditions appeared to be largely driven by biological activity distinct from the presence of dredge material. Isolation is an important driver in the degree to which extreme conditions can be generated so that a goal of beneficial use projects should be to minimize the degree of isolation that results from ponded water formed in dredge material placement areas.

The role of flushing was evaluated by considering measured variables during spring vs neap tidal conditions over the study period. Both pH and salinity values are higher and more variable at sample sites relative to control and reference sites in 2017 and 2018 compared to later in the study (Figure 63). This did not hold true for dissolved oxygen saturation, which remained variable throughout the study period, and for sites within the dredge material placement area as well as reference and control sites.

pH was significantly affected by flushing condition ($F_1 = 4.24$, $p < 0.05$); higher pH was detected in neap tide conditions (7.76 ± 0.03) than in spring tide conditions (7.66 ± 0.03). The interaction between water type (surface water vs. groundwater) and site type (control vs placement) also affected pH ($F_1 = 22.11$, $p < 0.001$); surface water pH was significantly higher at placement sites (8.54 ± 0.04) than at control sites (8.12 ± 0.05, $p < 0.0001$), while there was no significant difference in groundwater pH between site types.

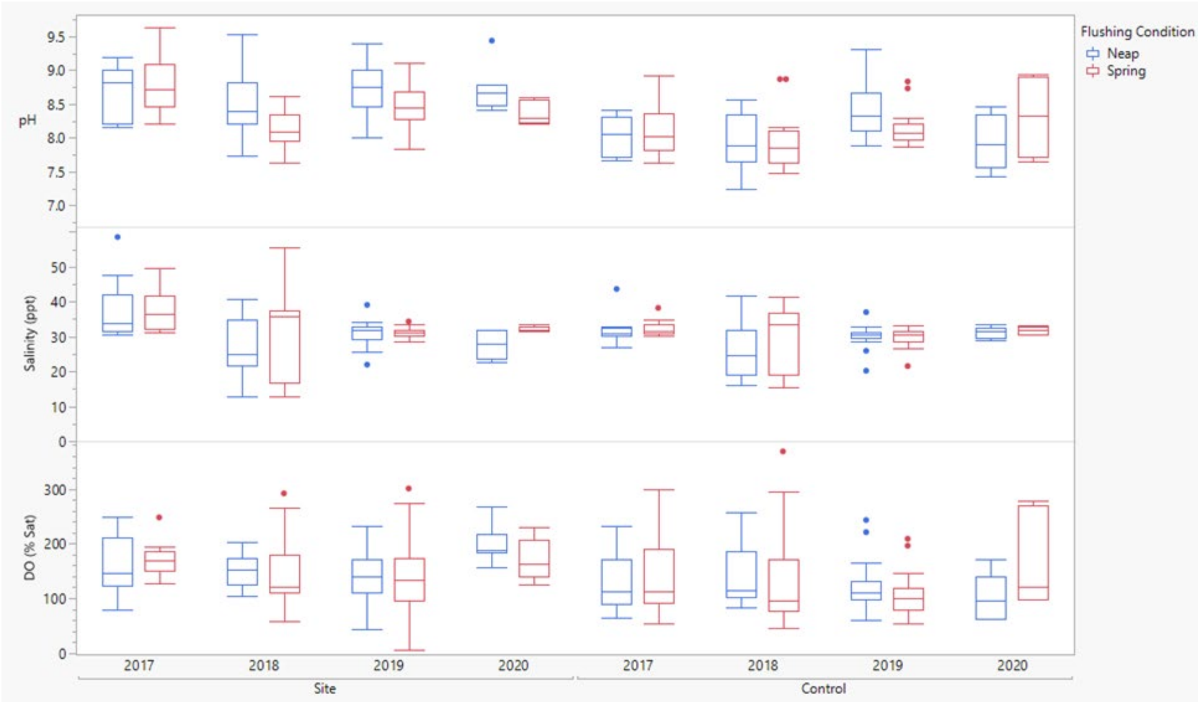


Figure 63. *Surface water pH, salinity and dissolved oxygen saturation (%) for reference/control and dredge material placement sites by year and flushing condition (spring vs neap) sampling periods.*

Groundwater Sampling Stations

Salinity and pH

The salinity of groundwater sampled from wells within the dredged material had higher overall salinities and wider ranges of salinity than groundwater sampled from control and reference wells (Figure 60). Wells and piezometers with the highest salinity (51.5 to 53.3, 46.3 to 45.1 ppt, respectively) were recorded from the P4-P6 cluster, which is located on the marsh plain above MHHW and differs from other well clusters in that it is not situated within the standing waters of a pool or panne. The lowest salinities were recorded in control and site wells and are especially concentrated around the 11/2018 sampling period with salinities as low as 17 ppt recorded at several stations. This was a spring tide sampling event associated with 5-day prior rainfall of nearly 2”. The estuary control site also showed very low salinity (16.8 ppt) at this time as did several surface water sites.

Salinity in groundwater (34.22 ± 0.45 ppt) was significantly higher than in surface water (30.63 ± 0.42 ppt, $F_1 = 34.25$, $p < 0.001$), and salinity at placement sites (33.72 ± 0.37 ppt) was significantly higher than at control sites (31.13 ± 0.49 ppt, $F_1 = 17.8$, $p < 0.001$). No significant impacts of flushing condition were detected for groundwater.

pH from groundwater sampled from wells located in the dredged material placement areas had similar mean pH to the control and reference wells (7.1, 7.1, and 7.0, respectively) but the range of pH for the groundwater sampling stations was more variable than the control and reference wells (Figure 60). Groundwater well P1-P3 Well 1 had a measured pH of 3.44 in June 2019 and 4.54 in June 2018 and well

P4-P6 Well 2 had measured pH of 5.71 and 5.91 both in June 2018. These are the lowest pH conditions recorded during the monitoring period and are lower than groundwater measurements in control and reference wells. The dissolved oxygen of all groundwater is low as expected (Figure 60).

The relationship between salinity and pH of groundwater sampled from wells within the dredge material was evaluated (Figure 64). For each piezometer and well, the depth below the surface of the screened interval was compared to the measured dredge material thickness at the sites. Sites were then categorized as to whether the screened interval was within the dredge material or below the dredge material in the underlying initial marsh, pool, or panne. The highest salinity and lowest pH (<6.5) occurred in the wells and piezometers that were within the dredge material (P1-P3 well 1, P3, P4-P6 Well 1, P4-P6 Well 2, P7-P9 Well 1, P9). This includes two instances of pH below 5. Sites with salinities >45 ppt are all from stations where the waters were drawn from wells/piezometers within the dredged material. The highest salinity measurements are concentrated in 2017 and 2018 samples and notably fewer in the 2019 and 2020 monitoring periods suggesting site evolution over time.

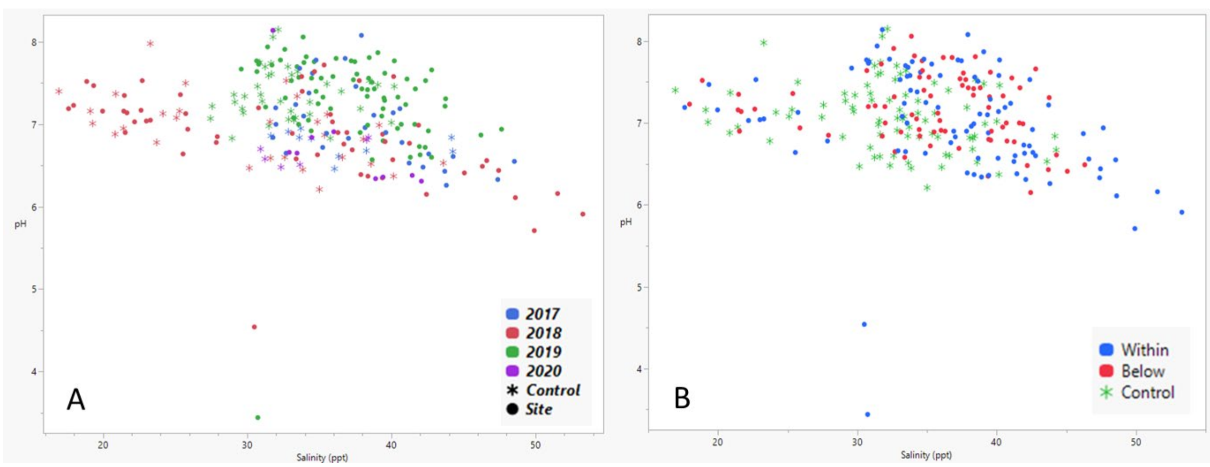


Figure 64. *A. pH and Salinity at Groundwater Control and Sampling Sites by Year, and B. pH and Salinity at Groundwater Control and Sampling Sites Identified by Location of Source Waters Relative to Dredged Materials.*

Samples that were drawn from below the dredge material (P1, P1-P3 well 2, P4, P6, P7) largely had higher pH (near 7 and above) and lower salinities (41 ppt and below). Surface sites adjacent to pools containing the wells and piezometer clusters (E2 and E4) displayed relatively high pH, further suggesting that the source of the lower pH in piezometers and wells with waters within the dredged materials was locally generated. Control and reference wells were predominantly comprised of waters with moderate pH and salinities relative to sites located in the placement areas (Figure 64). The control well and reference wells had mean pH of 7.1 and 7.0 and mean salinities of 32.8 and 31.7, respectively. Control well G2A on the marsh plain had relatively high salinities (44 ppt) and behaved as an outlier periodically.

These findings indicate that the dredge materials had relatively high salinities and were also likely generating acidic conditions. The very high surface water salinities recorded in isolated pools (e.g., D1) especially in 2017 when containment was still in place supports this interpretation.

Discussion

- Marsh pannes and pools display a wide range of surface water conditions that deviate from estuary controls and include wide ranges in pH, dissolved oxygen, and salinity into ranges that can be considered extreme. The degree of isolation and stagnation is an important driver of these conditions. Interior drainage of high salinity water generated by the dewatering of dredge materials is a likely cause of very high salinity surface waters especially since they were recorded early in the monitoring program. These highly variable conditions occurred in reference depressions as well as isolated ponded areas within the dredged material placement areas at Avalon indicating that the contribution to surface water chemistry from the dredged materials alone may be limited.
- The frequency of flushing is an important driver of surface water chemistry at dredge material placement sites. When sites were actively tidally flushed, surface water chemistry was more moderate. An initial goal of this monitoring program was to evaluate the effects of containment. However, only one monitoring event occurred prior to containment removal (March 2017). This monitoring event recorded the most extreme salinity conditions at the Avalon site (58.5 ppt). Overall, the most extreme surface water chemical conditions occurred in 2017 and 2018, with water chemistry parameters becoming more moderated over time. The limited data related to containment and the overall trend in more moderated water chemistry parameters makes it impossible to determine the effects of containment from this study.
- The presence of cyanobacterial mats within pools is an important driver of surface water chemistry and is likely responsible for elevated pH and oxygen supersaturation. Cyanobacterial mats occurred in abundance in the very shallow pools and depressions at the site, especially in the spring, summer, and fall. They did not occur in deeper natural pannes or where grazing epifauna were noted.
- Analysis of groundwater from dredge material sites revealed periods of high salinities and acidic conditions, especially among site wells and piezometers with water sourced from within dredged sediment. The range of pH measured in the groundwater wells was more variable than in the control and reference wells. Two groundwater wells within placement areas had measured pH values as low as 3.44 in June 2019 and 4.54 in June 2018 while a nearby well had measured pH of 5.72 and 5.91, both in June 2018. These are the lowest pH conditions recorded during the monitoring period. These conditions are lower than measured pH in control or reference wells and indicate that acidic conditions were being locally generated in the dredged sediments at least for a portion of the study period.
- This monitoring program did not document acidic surface water however, biologically produced alkaline pH may have masked those signals. The extremely reducing monosulfidic soils present in the dredge material are an important driver of soil chemistry. The effects of these reducing conditions were not apparent in surface waters. It is likely that cyanobacterial mats effectively sealed reducing conditions beneath the mats and also produced oxygen. In effect, the

biologically driven oxygenation of surface waters far exceeded any effects from reduced sediments, if any.

- Results of this study suggest that site water chemistry evolved over the study period and will continue to do so. This study documented that shallow, isolated pools have extreme water chemistry. These conditions can be naturally occurring; however, they occurred most frequently in the monitoring timeframe early in the study.

Appendices

Appendix A: Additional Vegetation Recovery Figures

Appendix B: Additional Avian Monitoring Figures and Data

Appendix C: Additional Nekton Data and Figures

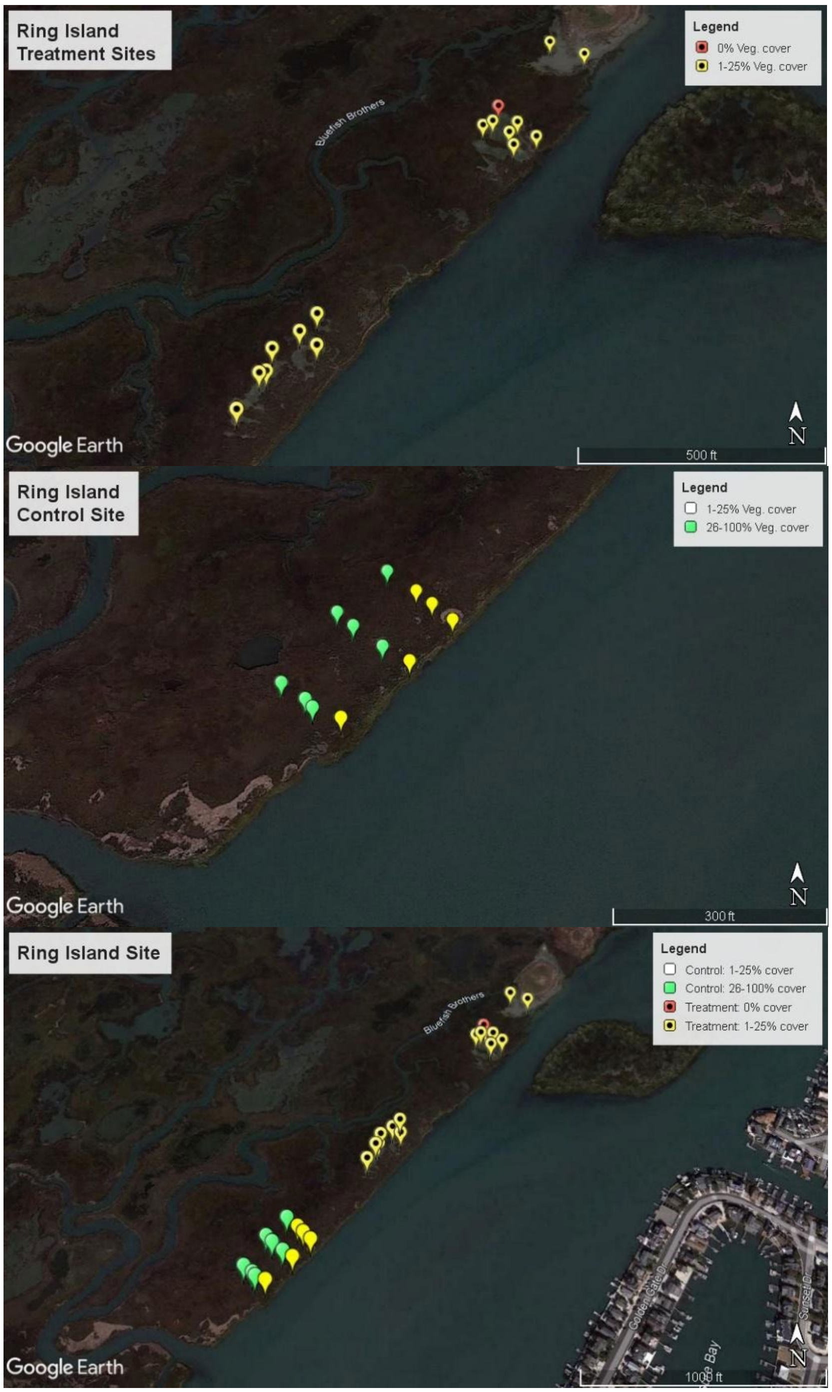
Appendix D: Monitoring Metrics, Sub-metrics, and Corresponding Units

Appendix E: References

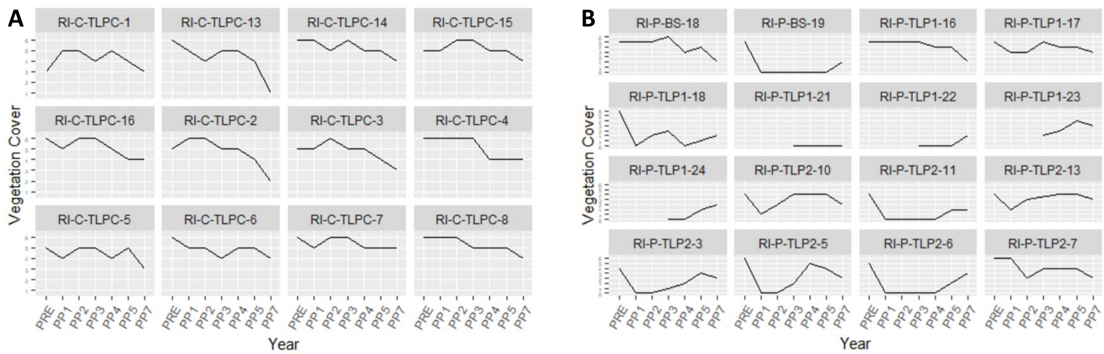
Appendix A

Additional Vegetation Recovery Figures

Ring Island



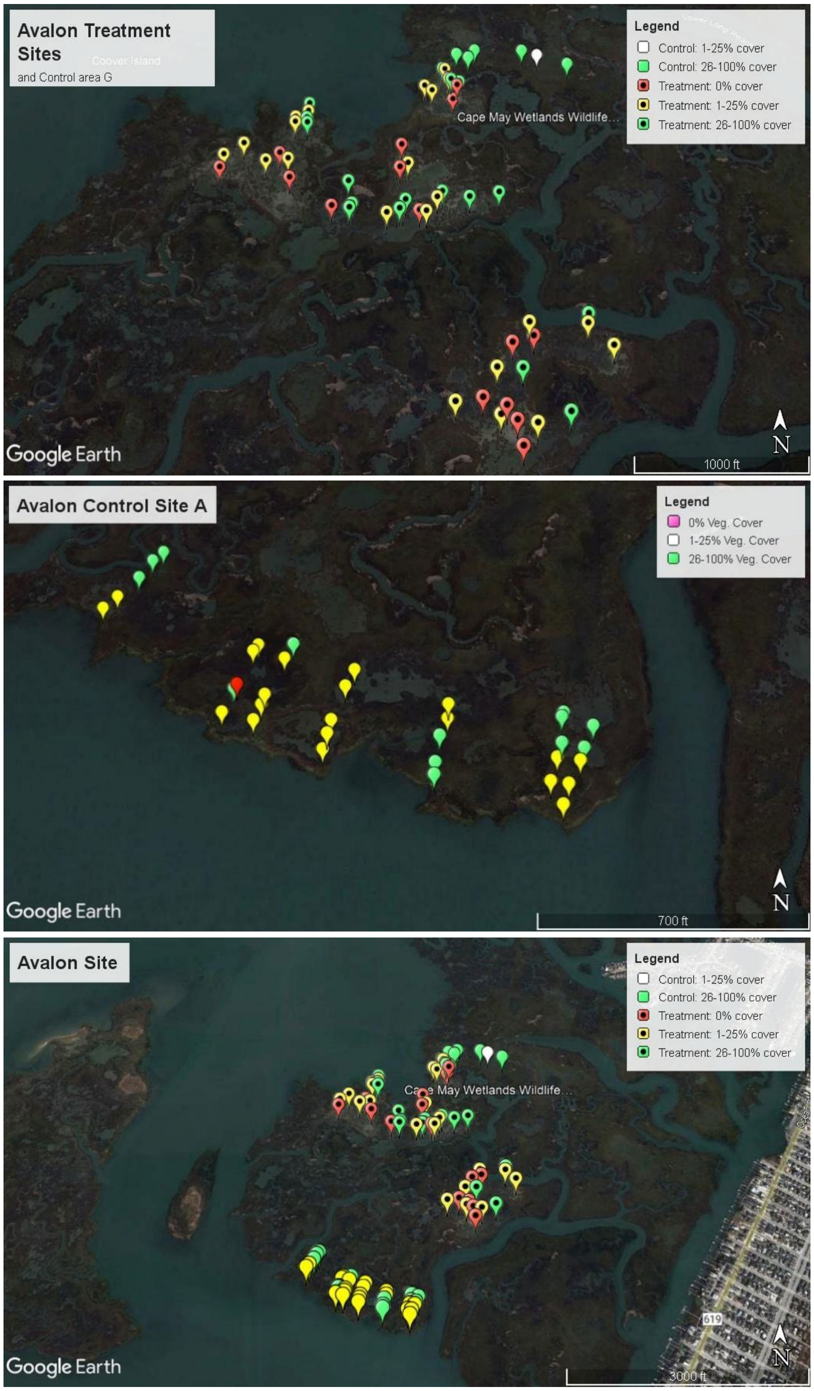
Map of plot recovery levels at Ring Island. Top map is a close-up of all Ring Island treatment sites. The center map is a close-up of the Ring Island control site, and the bottom map is the entirety of the Ring Island site in relation to the town of Stone Harbor and the back bay geography. Plots with black dots in the center of the icon are treatment plots, those without dots are control plots. Level of recovery is indicated by color, with 0% vegetation cover represented by red icons, 1-25% vegetation cover represented by yellow icons, and 26-100% vegetation cover represented by green icons.



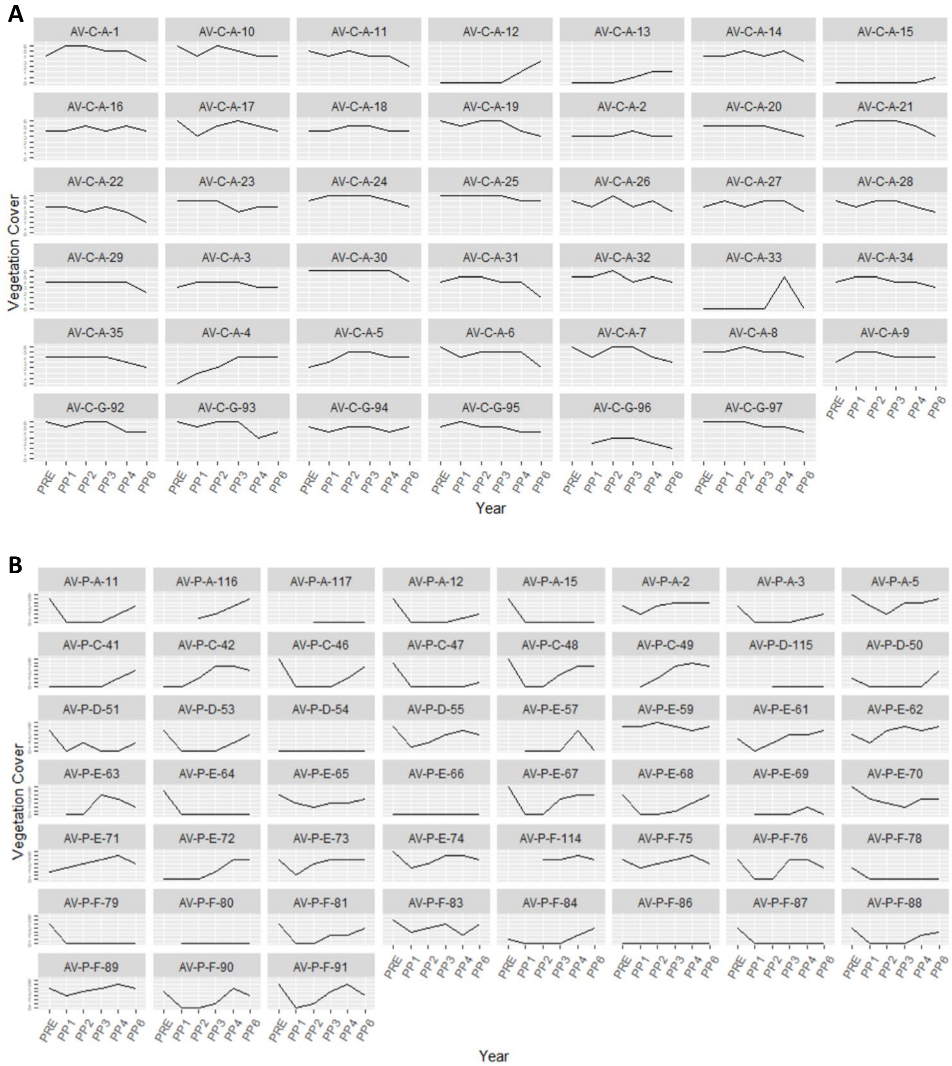
Vegetation percent cover change in individual A) control and B) placement plots at Ring Island over time.

Timeseries of plot 23 placement area 1 at Ring Island From left to right This was still predominantly bare inPP3 (left), the plot was initially colonized by Spartina alterniflora which gradually expanded into the plot from the edges in PP4 (middle) and PP5 (right).

Avalon

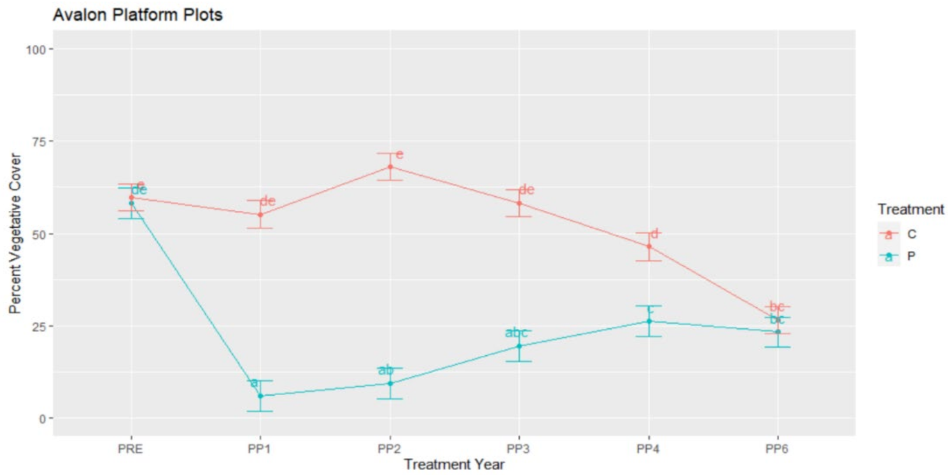


Map of plot recovery levels at Avalon. Top map is a close-up of all Avalon treatment sites, plus the control area G. The center map is a close-up of Avalon control site A, and the bottom map is the entirety of the Avalon site in relation to the town of Avalon and the back bay geography. Plots with black dots in the center of the icon are treatment plots, and those without dots are control plots. The level of recovery is indicated by color, with 0% vegetation cover represented by red icons, 1-25% vegetation cover represented by yellow icons, and 26-100% vegetation cover represented by green icons.

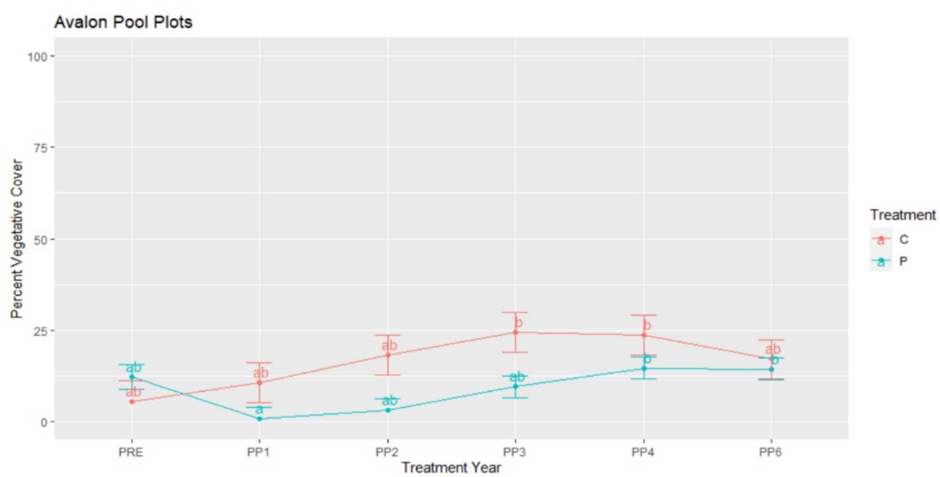


Vegetation percent cover change in individual A) control and B) treatment plots at Avalon over time.

Timeseries of plot #71 in Avalon placement area E. This plot started as a pool prior to placement PRE (left). The plot was considered bare the first 2 years post placement. In PP2 (middle left) the plot was colonized by Salicornia sp. In PP3(middle right) Spartina alterniflora began to establish and by PP4 (far right), the plot is fully vegetated by Spartina alterniflora.

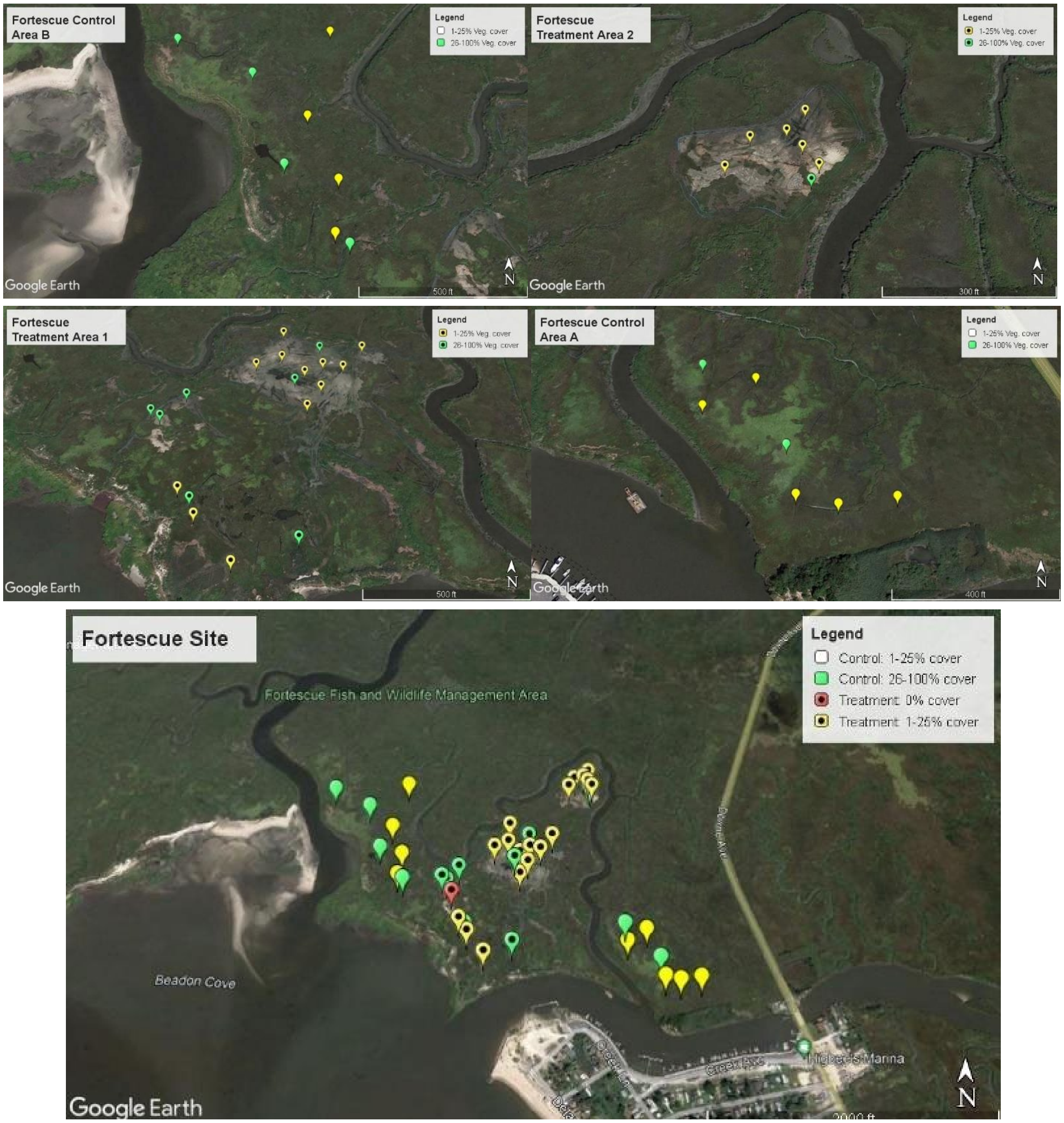


Percent cover by plant at Avalon in platform plots in both Control (C) and Placement Areas (P) overtime. Data presented are means ± one standard error.

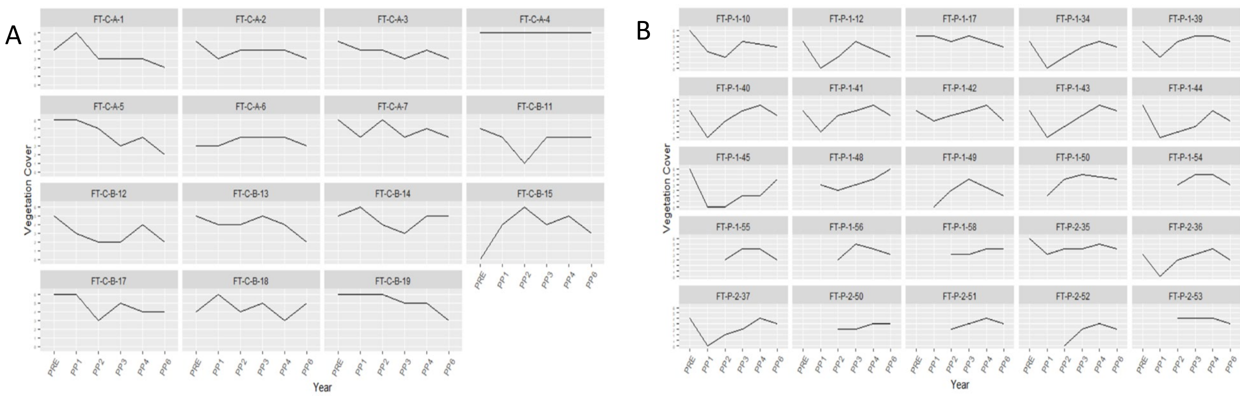


Percent cover by plant at Avalon in pool plots in both Control (C) and Placement Areas (P) overtime. Data presented are means ± one standard error.

Fortescue



Map of plot recovery levels at Fortescue. The top four maps are close-ups of treatment and control sites at Fortescue. The bottom map is the entirety of the Fortescue site in relation to the marina, local development, and the nearby bay geography. Plots with black dots in the center of the icon are treatment plots, those without dots are control plots. The level of recovery is indicated by color, with 0% vegetation cover represented by red icons, 1-25% vegetation cover represented by yellow icons, and 26-100% vegetation cover represented by green icons.



Vegetation percent cover change in individual A) control and B) treatment plots at Fortescue over time.

Successional photos taken from a permanently established photo point at the Fortescue project site in 2016 and 2017 (PP1 and PP2, respectively).

Timeseries of plot #37 in Fortescue placement area 2. From left to right as described: the plot was completely bare following placement In PP1. Some vegetation began expanding into one corner from nearby existing vegetation in PP2., In PP3 the plot was sparsely vegetated throughout most of the area with PP4 showing much denser vegetation at 38% cover.

Appendix B

Additional Avian Monitoring Figures

Avalon

Map of monitoring locations for avian counts in Avalon. All birds within the red circles were considered part of the analysis. Site names are coded for location (e.g., A = Avalon), the timing of placement (i.e., before or after; B or A, respectively), site descriptor (i.e., control, impact, or demo; C, I, or D, respectively), and cardinal direction (e.g., N = north).

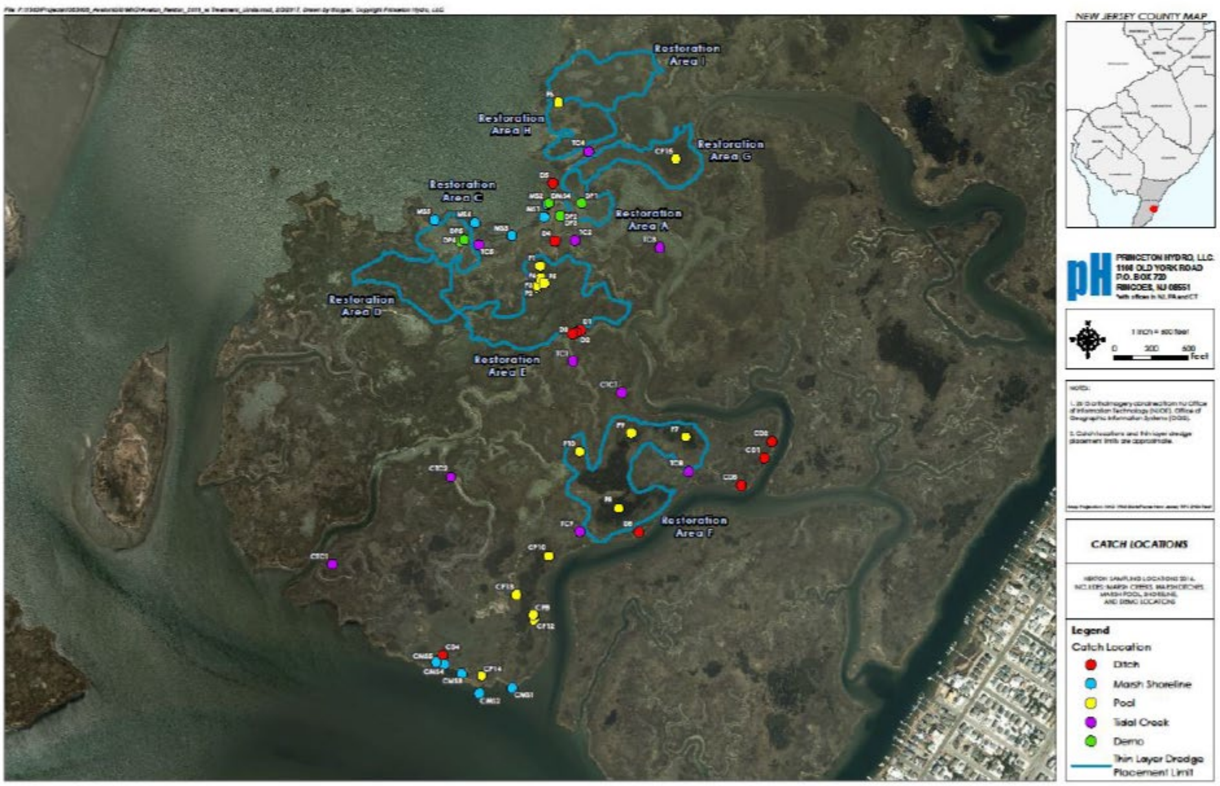
Fortescue

Map of monitoring locations for avian counts in Fortescue. All birds within the red circles were considered part of the analysis. Site names are coded for location (e.g., F = Fortescue), the timing of placement (i.e., before or after; B or A, respectively), site descriptor (i.e., control or impact; C or I, respectively), and cardinal direction (e.g., N = north).

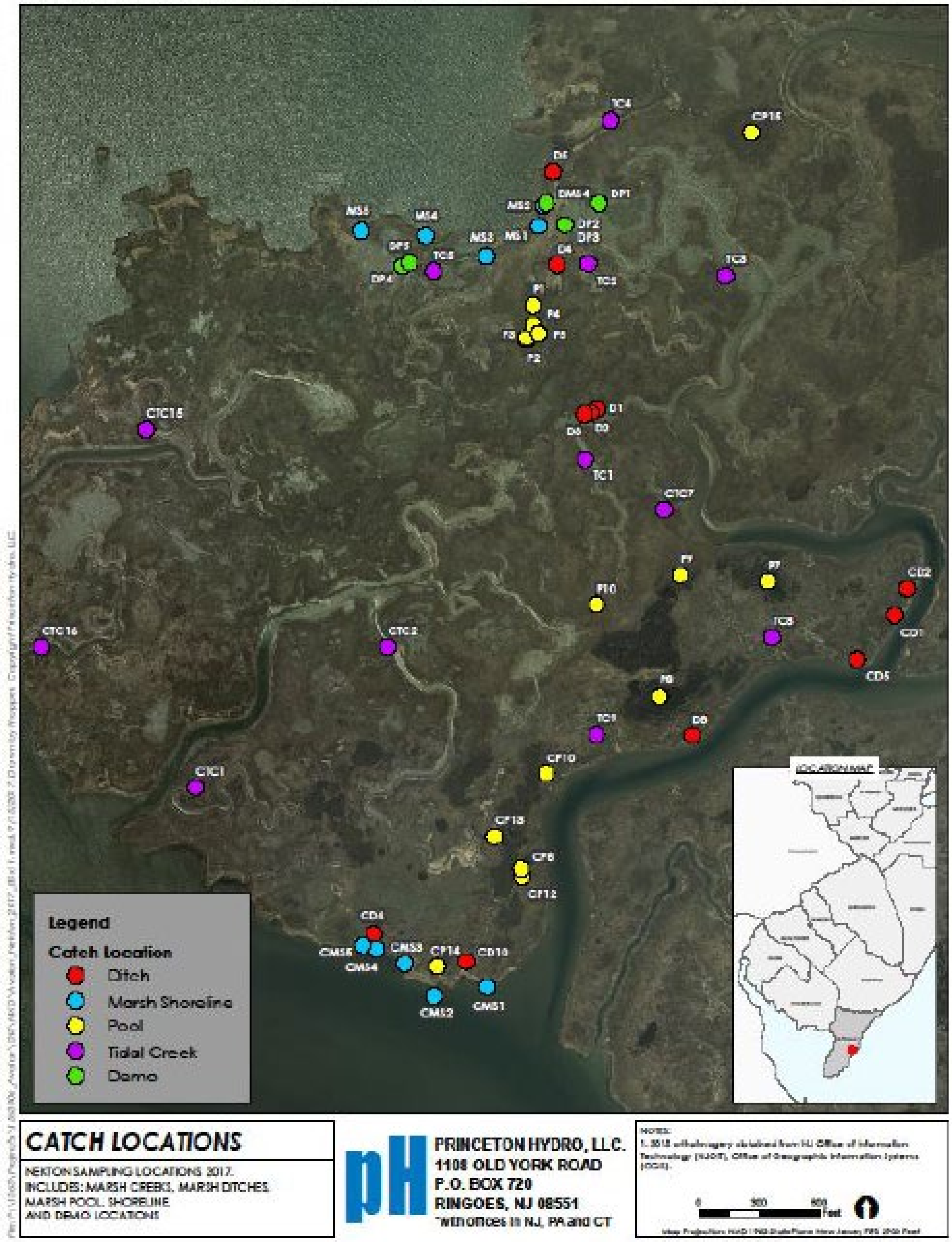
Appendix C

Additional Nekton Data and Figures

Avalon



Nekton by treatment and habitat sampling locations at Avalon, New Jersey 2016. Maps created by Thomas Hopper of Princeton Hydro, LLC.

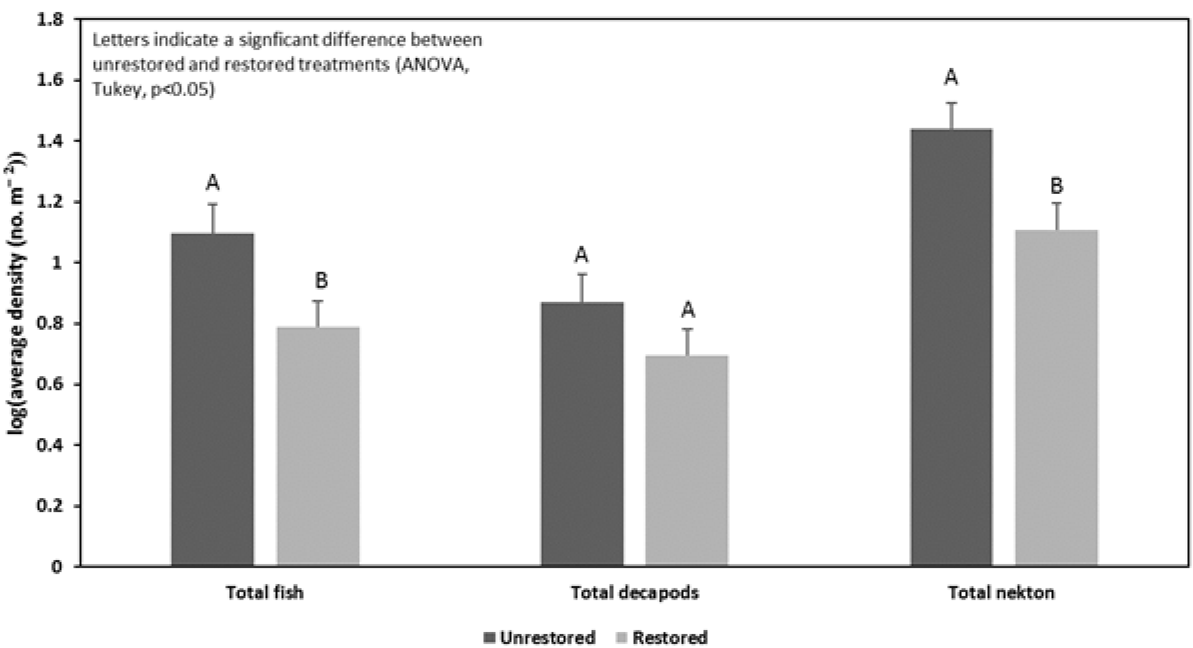


Nekton by treatment and habitat sampling locations at Avalon, New Jersey 2017.

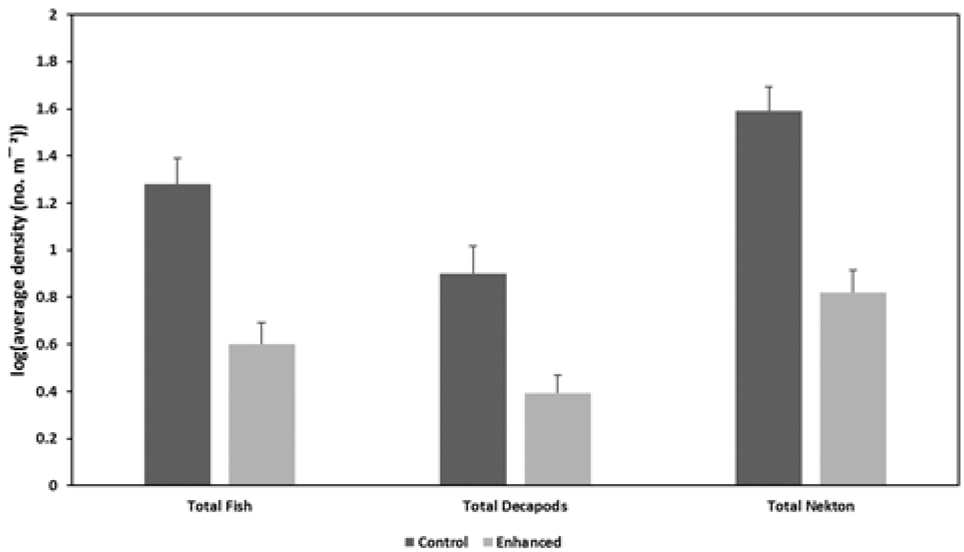
Total nekton abundance in tidal creeks, pools, subtidal creeks, and the shoreline edge at Avalon, NJ 2017

Species (Common Name)	2017 Avalon							
	Control				Enhanced			
	Tidal Creek n=5/5	Pools n=6/6	Shoreline Edge n=5/5	Subtidal Creek n=5/4	Tidal Creek n=7/7	Pools n=9/9	Shoreline Edge n=5/5	Subtidal Creek n=6/6
Fundulus heteroclitus (mummichog)	98	16	85	203	99	2		13
Fundulus majalis (striped killifish)	189	51	66	284	711	40	2	23
Lucania parva (rainwater killifish)		186	2			5		6
Menidia menidia (Atlantic silverside)	139	44	27	66	47	3	16	5
Palaemonetes pugio (daggerblade grass shrimp)	482	65	600	21	443		34	28
Carcinus maenas (green crab)								
Crangon septemspinosa (sevenspine bay shrimp)							5	
Callinectes sapidus (blue crab)	37	5	61	7	17	2	25	
Panopeus herbstii (Atlantic mud crab)								
Cyprinodon variegatus (sheepshead minnow)	3	400		150	6	59		
Gobiosoma bosc (naked goby)	2		14		1		6	
Mugil cephalus (striped mullet)								
Uca pugnax (Atlantic marsh fiddler crab)	1			3				4
Micropogonias undulatus (Atlantic Croaker)								
Brevootia tyrannus (Atlantic menhaden)	1		5					
Syngnathus fuscus (northern pipefish)			1					
Pogonias cromis (black drum)								
Eurypanopeus depressus (flatback mud crab)	3							

n= number of sampling sites within each habitat during Event 1/Event 2



2015-2016 average nekton density for fishes, decapods, and nekton at Avalon



2017 average nekton density for fishes, decapods, and nekton at Avalon

Average nekton length (length mm ± standard error) in tidal creeks, pools, subtidal creeks, and the shoreline edge at Avalon, NJ 2016.

Species (Common Name)	2016 Avalon							
	Unrestored				Restored			
	Tidal Creek n=6	Pools n=12	Shoreline Edge n=10	Subtidal Creek n=8	Tidal Creek n=14	Pools n=18	Shoreline Edge n=10	Subtidal Creek n=12
Fundulus heteroclitus (mummichog)	37.80 ±5.96	26.36± 1.66	35.93± 3.31	37.43± 2.79	43.61 ±4.96	43.00 ±0	27.80± 0	42.16± 2.89
Fundulus majalis (striped killifish)	43.28 ±6.75	26.96± 1.98	46.70± 4.26	47.18± 9.97	47.37 ±6.45	28.35 ±3.95	34.50± 0	39.86± 3.59
Lucania prava (rainwater killifish)		20.39± 1.11	22.00± 0			16.47 ±4.62		21.00± 0
Menidia menidia (Atlantic silverside)	25.33 ±2.67	31.68± 3.38	31.00± 0		32.45 ±3.04		35.75± 1.75	32.00± 0
Palaemonetes pugio (daggerblade grass shrimp)	26.88 ±1.51	22.9± 2.39	28.67± 1.94	35.00± 4.51	24.57 ±1.63	14.00 ±0	25.17± 1.60	28.4 ±1.72
Carcinus maenas (green crab)								
Crangon septemspinosa (sevenspine bay shrimp)	25.50 ±0.5				27.00± 0			
Callinectes sapidus (blue crab)	22.60 ±6.59	63.5± 9.33	12.50± 0.50	14.00 ±0	24.67 ±10.65	10.33 ±1.67	35.67± 20.80	
Panopeus herbstii (Atlantic mud crab)		12.00± 0	14.80 ±1.88	23.50± 6.50	8.00 ±0			18.33± 0.33
Cyprinodon variegatus (sheepshead minnow)	36.50± 0	25.22± 1.65	32.00± 0		37.13 ±2.07	32.47 ±1.60		40.30± 0
Gobiosoma bosc (naked goby)	41.00± 0		33.00± 4.16		29.50± 0		36.00 ±0	29.00± 0
Mugil cephalus (striped mullet)					40.00± 0			
Uca pugnax (Atlantic marsh fiddler crab)								14.00± 0
Brevootia tyrannus (Atlantic menhaden)	88.80 ±0							
Micropogonias undulatus (Atlantic Croaker)	62.00± 0							
Syngnathus fuscus (northern pipefish)			76.5± 0					
Pogonias cromis (black drum)								

n= number of sampling sites within each habitat

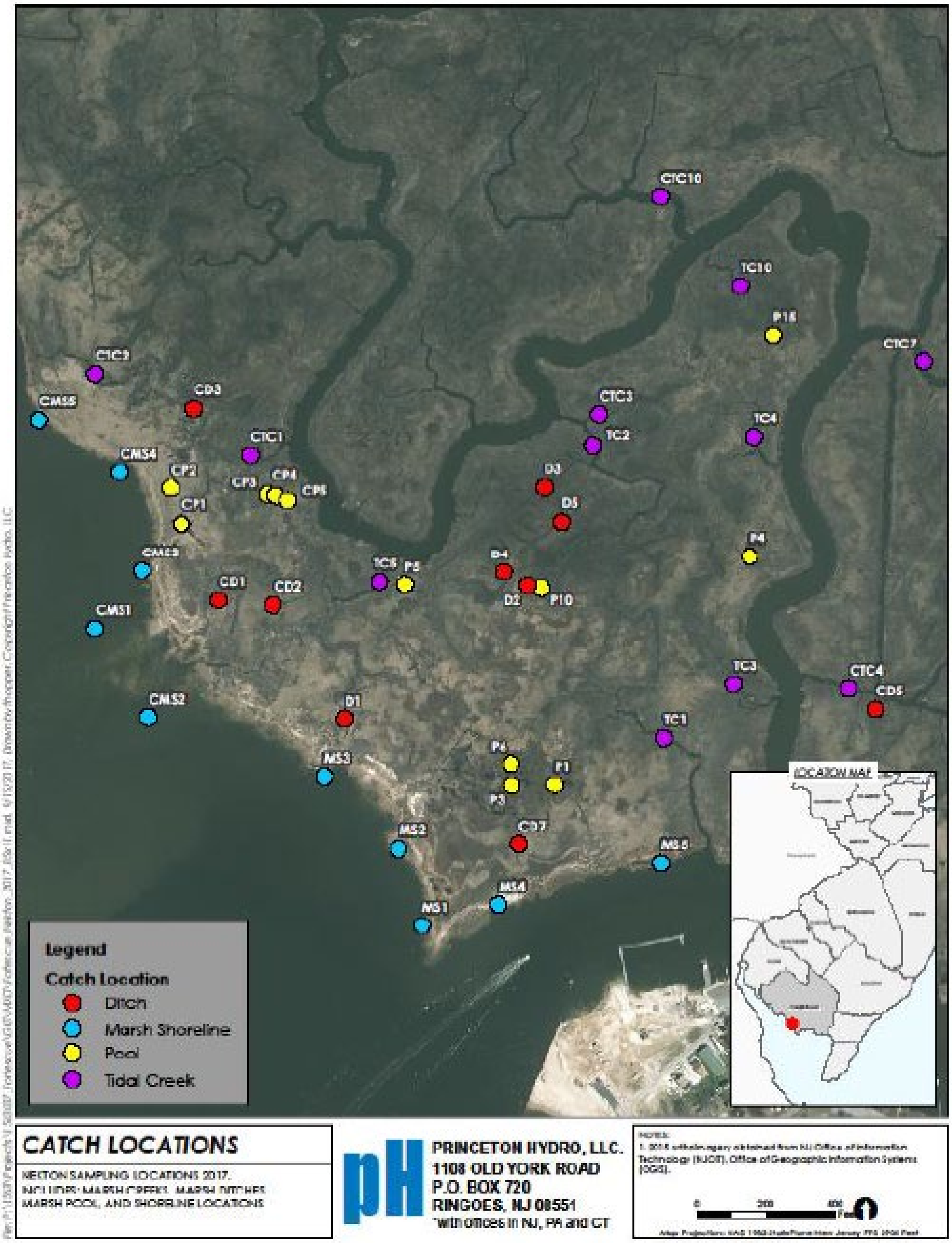
Average nekton length (length mm ± standard error) in tidal creeks, pools, subtidal creeks, and the shoreline edge at Avalon, NJ 2017.

Species (Common Name)	2017 Avalon							
	Control				Enhanced			
	Tidal Creek n=5/5	Pools n=6/6	Shoreline Edge n=5/5	Ditch n=5/4	Tidal Creek n=7/7	Pools n=9/9	Shoreline Edge n=5/5	Ditch n=6/6
Fundulus heteroclitus (mummichog)	38.34 ± 1.01	35.46 ± 1.62	26.41 ± 2.21	44.78 ± 1.14	38.41 ± 2.13	10.50 ± 0		33.44 ± 4.12
Fundulus majalis (striped killifish)	41.48 ± 3.65	28.93 ± 1.36	27.00 ± 3.32	34.82 ± 2.79	33.26 ± 2.07	34.40 ± 5.89	27.00 ± 5.00	33.54 ± 2.70
Lucania parva (rainwater killifish)		19.38 ± 1.09	17.00 ± 0			19.40 ± 6.40		22.70 ± 0
Menidia menidia (Atlantic silverside)	34.85 ± 3.10	28.10 ± 2.93	31.39 ± 4.53	33.98 ± 2.55	32.54 ± 3.71	33.50 ± 4.50	32.38 ± 7.95	33.33 ± 3.33
Palaemonetes pugio (daggerblade grass shrimp)	25.67 ± 1.38	22.60 ± 1.21	27.78 ± 1.14	27.60 ± 1.75	27.25 ± 1.11		25.33 ± 4.60	26.20 ± 0.80
Carcinus maenas (green crab)								
Crangon septemspinosa (sevenspine bay shrimp)							20.67 ± 2.73	
Callinectes sapidus (blue crab)	13.00 ± 2.08	56.00 ± 18.27	13.25 ± 0.88	13.00 ± 4.00	21.60 ± 7.28	8.50 ± 0.50	32.43 ± 18.22	
Panopeus herbstii (Atlantic mud crab)								
Cyprinodon variegatus (sheepshead minnow)	31.25 ± 2.75	23.27 ± 1.05		33.50 ± 0	29.63 ± 1.28	19.75 ± 3.08		
Gobiosoma bosc (naked goby)	38.00 ± 0		29.64 ± 1.42		28.00 ± 0		28.75 ± 10.25	
Mugil cephalus (striped mullet)								
Uca pugnax (Atlantic marsh fiddler crab)	19.00 ± 0			16.00 ± 4.04				18.33 ± 2.03
Brevoortia tyrannus (Atlantic menhaden)	27.00 ± 0		14.77 ± 2.43					
Micropogonias undulatus (Atlantic Croaker)								
Syngnathus fuscus (northern pipefish)	128.00 ± 0							
Pogonias cromis (black drum)								
Eurypanopeus depressus (flatback mud crab)	17.30 ± 0							

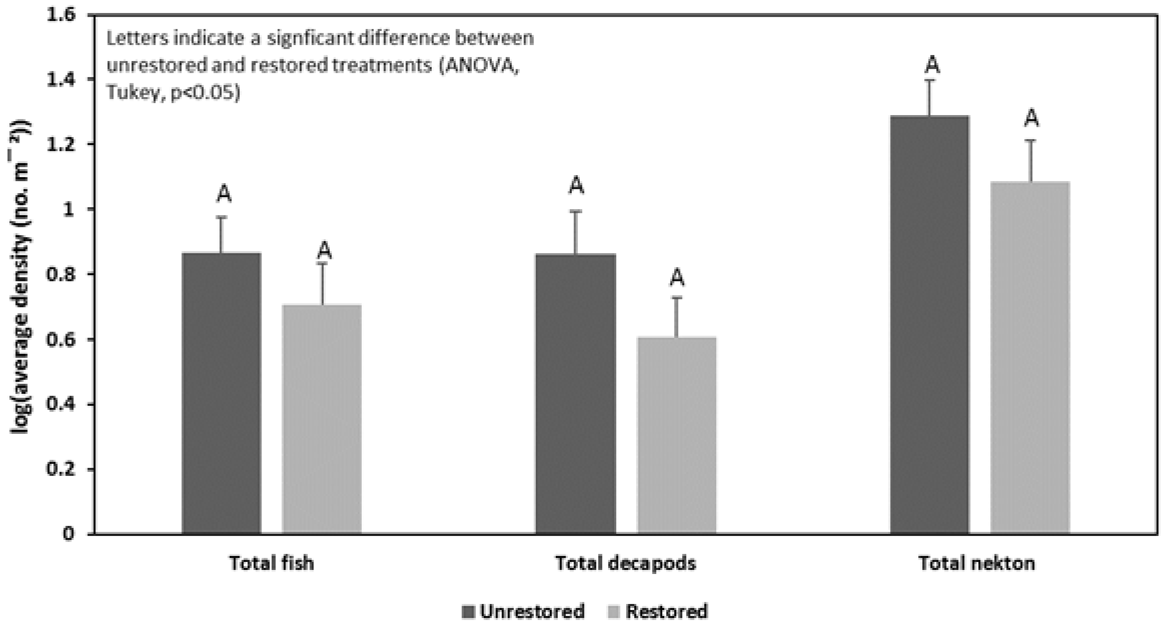
n= number of sampling sites within each habitat

Fortescue

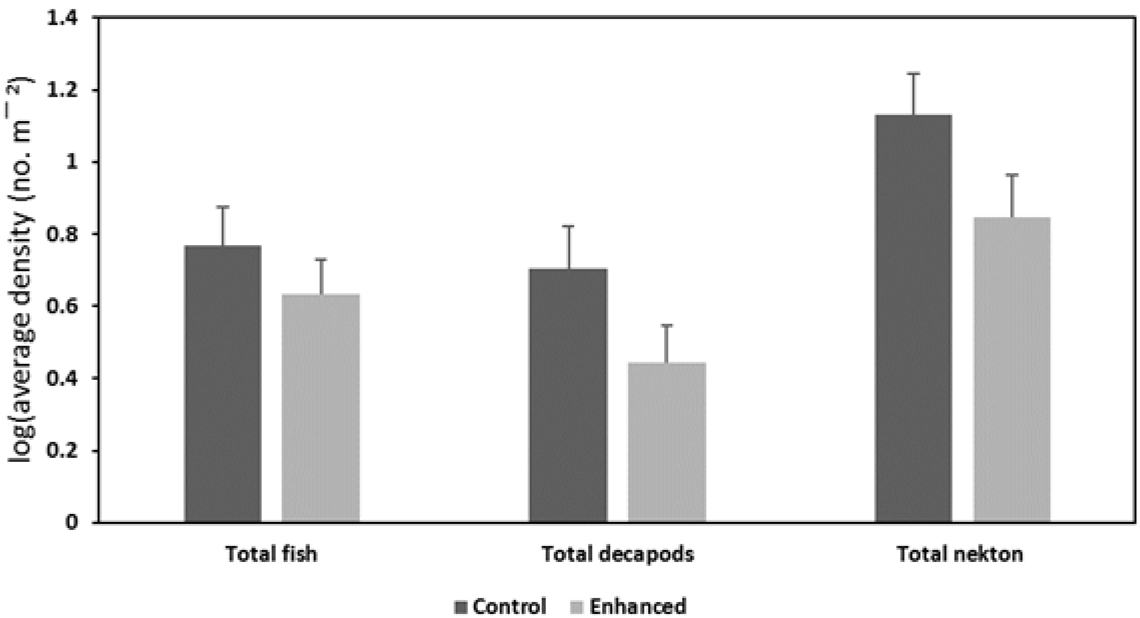
Nekton by treatment and habitat sampling locations at Fortescue, New Jersey 2016. Maps created by Thomas Hopper of Princeton Hydro, LLC.



Nekton by treatment and habitat sampling locations at Avalon, New Jersey 2017.



2016 average nekton density for fishes, decapods, and nekton at Fortescue



2017 average nekton density for fishes, decapods, and nekton at Fortescue

Average nekton length (length mm ± standard error) in tidal creeks, pools, subtidal creeks, and the shoreline edge at Fortescue, NJ 2016.

Species (Common Name)	2016 Fortescue							
	Unrestored				Restored			
	Tidal Creek	Pools	Shoreline Edge	Ditch	Tidal Creek	Pools	Shoreline Edge	Ditch
	n=5	n=10	n=9	n=10	n=5	n=10	n=10	n=8
Fundulus heteroclitus (mummichog)	39.03 ± 2.43	35.16 ± 3.43		38.22 ± 3.36	49.74 ± 5.80	36.58 ± 3.01	89.00 ± 0	37.93 ± 0.52
Fundulus majalis (striped killifish)	39.47 ± 2.76	31.90 ± 2.90	52.00 ± 0	35.52 ± 2.74	45.85 ± 7.95	33.73 ± 2.28	74.30 ± 0	31.80 ± 2.52
Lucania prava (rainwater killifish)		24.21 ± 1.45		25 ± 0		28.87 ± 1.85		29.75 ± 1.55
Menidia menidia (Atlantic silverside)	40.00 ± 0	38.04 ± 2.84	61.53 ± 6.14		37.75 ± 3.08		28.00 ± 0	
Palaemonetes pugio (daggerblade grass shrimp)	21.75 ± 1.49	19.71 ± 3.44	24.40 ± 2.38	20.00 ± 1.92	29.67 ± 2.67	16.33 ± 1.58	26.17 ± 1.08	19.00 ± 0
Crangon septemspinosa (sevenspine bay shrimp)			29.00 ± 0			12.50 ± 0.50	36.00 ± 1.00	
Callinectes sapidus (blue crab)	11.50 ± 0.50	29.80 ± 4.53	14.00 ± 4.92	33.00 ± 0	17.00 ± 1.00	38.5 ± 15.37	24.80 ± 9.02	
Panopeus herbstii (Atlantic mud crab)	5.00 ± 0			8.00 ± 0				8.00 ± 0
Cyprinodon variegatus (sheepshead minnow)		31.80 ± 1.67		29.83 ± 5.63		32.13 ± 0.30		35.10 ± 0
Gobiosoma bosc (naked goby)			19.00 ± 8.00					
Uca pugnax (Atlantic marsh fiddler crab)	22.00 ± 0			39.00 ± 6.00				
Micropogonias undulatus (Atlantic croaker)								
Trinectes maculatus (hogchoker)			25.50 ± 2.50					

n= number of sampling sites within each habitat

Average nekton length (length mm ± standard error) in tidal creeks, pools, subtidal creeks, and the shoreline edge at Fortescue, NJ 2017.

Species (Common Name)	2017 Fortescue							
	Control				Enhanced			
	Tidal Creek	Pools	Shoreline Edge	Ditch	Tidal Creek	Pools	Shoreline Edge	Ditch
	n=5/6	n=5/5	n=5/5	n=5/5	n=7/6	n=7/7	n=5/5	n=5/5
Fundulus heteroclitus (mummichog)	39.34 ± 2.08	31.05 ± 2.22		41.18 ± 5.00	46.88 ± 2.47	37.29 ± 1.80		31.15 ± 1.85
Fundulus majalis (striped killifish)	38.37 ± 1.37	29.11 ± 1.73		33.18 ± 1.42	38.58 ± 4.70	29.59 ± 2.18		28.98 ± 2.15
Lucania prava (rainwater killifish)		26.42 ± 1.42		23.75 ± 1.75	23.37 ± 2.30			21.50 ± 0.50
Menidia menidia (Atlantic silverside)	49.18 ± 6.00	36.45 ± 5.67	41.67 ± 9.39		36.79 ± 1.55		24.71 ± 0	
Palaemonetes pugio (daggerblade grass shrimp)	25.25 ± 0.86	21.67 ± 1.75	30.83 ± 1.62		28.00 ± 1.40	20.80 ± 2.40	28.50 ± 0.50	25.00 ± 0
Crangon septemspinosa (sevenspine bay shrimp)								
Callinectes sapidus (blue crab)	45.75 ± 17.82	39.33 ± 15.96	31.40 ± 15.92		32.00 ± 20.00			
Panopeus herbstii (Atlantic mud crab)					4.00 ± 0			
Cyprinodon variegatus (sheepshead minnow)		35.08 ± 2.72		29.00 ± 0	34.00 ± 3.00	33.08 ± 0.66		22.00 ± 0
Gobiosoma bosc (naked goby)		22.00 ± 0						
Uca pugnax (Atlantic marsh fiddler crab)	13.00 ± 0	32.50 ± 0		15.00 ± 1.00				
Micropogonias undulatus (Atlantic Croaker)								
Brevoortia tyrannus (Atlantic menhaden)								
Syngnathus fuscus (northern pipefish)								
Trinectes maculatus (hogchoker)								
Pogonias cromis (black drum)								

n= number of sampling sites within each habitat during Event 1/Event 2

Appendix D

Monitoring Metrics, Sub-metrics, and Corresponding Units

Metric	Sub-metrics	Units
Vegetation in permanent plots	Average stem height of dominant plant species	cm
Vegetation in permanent plots	Percent cover of vegetation (total and by cover types)	%
Vegetation in permanent plots	Species richness	Names and count
Vegetation	Above and below ground biomass	g (wet weight)/m^2; g (dry weight)/m^2
Salinity in permanent plots		ppt
Epifaunal macroinvertebrates in permanent plots	Species richness; abundance; density	NA; NA; classes
Bearing capacity in permanent plots	Depth of penetration	cm
Depth of placement in permanent plots		cm
Habitat changes over time (drone photos and orthophotos from GTA elevation surveys)		Acres or %
SETs and marker horizons	Elevation	NAVD 88
SETs and marker horizons	Elevation	mm
SETs and marker horizons	Shallow subsidence	mm
SETs and marker horizons	Net accretion	mm
Benthic infauna	Species diversity	Polychaetes to species level, crustaceans and mollusks to order level, count all nematodes, count all oligochaetes
Benthic infauna	Species density	Average abundance per core (3.8 cm diameter cores, top 3 cm)
Soil properties	Sediment grain size	% sand, silt, and clay
Soil properties	Organic matter	%
Soil properties	Soil moisture	mL H_20 per g_{soil}
Soil properties	Bulk density (without roots)	g/cm^3
Soil properties	Total volatile solids	%
Soil properties	Carbon	%
Soil properties	Nitrogen	%
Soil properties	Phosphorus	%
Soil properties	Sulfur	%

Soil properties	Microbial biomass carbon/nitrogen; potentially mineralizable nitrogen; extractable nitrate, ammonium, and phosphorus analysis	
Topography & Elevation		NAVD 88
Horizontal extent of placement		NAD 83
Surface Water Elevation & Tide Range Monitoring	Mean Higher High Water (MHHW) Elevation	NAVD 88
Surface Water Elevation & Tide Range Monitoring	Mean High Water (MHW) Elevation	NAVD 88
Surface Water Elevation & Tide Range Monitoring	Mid Tide Line (MTL) Elevation	NAVD 88
Surface Water Elevation & Tide Range Monitoring	Mean Low Water (MLW) Elevation	NAVD 88
Surface Water Elevation & Tide Range Monitoring	Mean Lower Low Water (MLLW) Elevation	NAVD 88
Surface Water Elevation & Tide Range Monitoring	Hydroperiod or tidal inundation depth/duration	
Nekton	Taxa richness	
Nekton	Composition	
Nekton	Species density	Number of individuals/m^2
Nekton	*Fundulus spp*. length	mm
In situ water quality	Temperature	°C
In situ water quality	Salinity/conductance	ppt
In situ water quality	Dissolved oxygen (concentration and percent saturation)	mg/L; %
In situ water quality	pH	
In situ water quality	Clarity/turbidity	
In situ water quality	Water depth	cm
Avian	Species richness	Number of species
Avian	Species abundance	Number of individuals per species
Avian	Species age	Classes
Avian	Species behavior	*e.g.,* nesting, loafing, foraging
Avian	Productivity	Number of fledglings per nesting pair
Avian	Number of nests	
Avian	Hatch success	Proportion of nests that hatch at least one egg
Avian	Fledging success	Proportion of nests that fledge at least one chick
Monthly observations	Fixed photo points	
Monthly observations	Depth of dredged material at the permanent depth stakes	cm

Monthly observations	Vegetation recovery/recruitment, vegetation die-off/notable absence, containment problems, sediment observations, instances of pooling, wildlife observations, planting success, planting failure, other	
Hydrology	Tidal datum	NAVD 88

Appendix E

References

Addink, E. and Coillie, F.V. 2010. *Object-based Image Analysis*. GIM International. Object-based Image Analysis | GIM International (gim-international.com)

Barbier, E.B., Hacker, S.D., Kennedy, C., Koch, E.W., Stier, A.C. and Silliman, B.R., 2011. The value of estuarine and coastal ecosystem services. *Ecological monographs*, 81.2: 169-193.

Bertness, M.D. and Ellison, A.M., 1987. Determinants of pattern in a New England salt marsh plant community. *Ecological monographs*, *57*(2), pp.129-147.

Bertness, M.D. and Miller, T., 1984. The distribution and dynamics of *Uca pugnax* (Smith) burrows in a New England salt marsh. *Journal of Experimental Marine Biology and Ecology*, *83*(3), pp.211-237.

Cahoon, D.R., 2015. Estimating relative sea-level rise and submergence potential at a coastal wetland. *Estuaries and Coasts*, *38*(3), pp.1077-1084.

Congalton, R.G. and Green, K., 2019. *Assessing the accuracy of remotely sensed data: principles and practices*. CRC press.

Congalton, R. G., Green, K., & Tukman, M. 2017. *Imagery and GIS: best practices for extracting information from imagery.* Esri Press.

Environmental Systems Research Institute (ESRI). (2016). ArcGIS Release 10.4. Redlands, CA.

Environmental Systems Research Institute (ESRI). (2020). ArcGIS Pro Release 2.6. Redlands, CA.

Environmental Systems Research Institute (ESRI). (2021). ArcGIS Pro Release 2.7.3. Redlands, CA.

Environmental Systems Research Institute (ESRI). "Segmentation". Segmentation-ArcGIS Pro | Documentation. https://pro.arcgis.com/en/pro-app/latest/help/analysis/image-analyst/segmentation.htm

Ferencz, A., Gruver, M., & McKenna, K. (2017). *HAZUS analysis estimating reductions in flood damage costs derived from salt marsh habitats fronting development using HAZUS at Avalon and Fortescue, New Jersey*. Stockton University Coastal Research Center.

Haaf, L., Watson, E.B., Elsey-Quirk, T., Raper, K., Padeletti, A., Maxwell-Doyle, M., Kreeger, D., and Velinsky, D.J., 2022. Sediment accumulation, elevation change, and the vulnerability of tidal marshes in the Delaware Estuary and Barnegat Bay to accelerated sea level rise. *Estuaries and Coasts*, *45*(2), pp.413-427.

Hartig, E.K., Gornitz, V., Kolker, A., Mushacke, F., and Fallon, D., 2002. Anthropogenic and climate-change impacts on salt marshes of Jamaica Bay, New York City. *Wetlands*, *22*(1), pp.71-89.

Howes, B.L., Dacey, J.W.H., and Goehringer, D.D., 1986. "Factors controlling the growth form of Spartina alterniflora: feedbacks between above-ground production, sediment oxidation, nitrogen and salinity." *The Journal of Ecology*, pp. 881-898.

James-Pirri, M.J., Swanson, J.L., Roman, C.T., Ginsberg, H.S., and Heltshe, J.F., 2014. Ecological thresholds for salt marsh nekton and vegetation communities. In *Application of threshold concepts in natural resource decision making* (pp.291-319). Springer, New York, NY.

Koller, M. and Stahel, W.A., 2011. Sharpening Wald-type inference in robust regression for small samples. *Computational Statistics and Data Analysis*, *55*(8), pp. 2504-2515. https://doi.org/10.1016/j.csda.2011.02.014

Litvin, S.Y. and Weinstein, M.P., 2003. Life history strategies of estuarine nekton: the role of marsh macrophytes, benthic microalgae, and phytoplankton in the trophic spectrum. *Estuaries and Coasts* 26.2: 552-562.

Maechler M., Rousseeuw P., Croux C., Todorov V,. Ruckstuhl A., Salibian-Barrera M., Verbeke T., Koller M., Conceicao E.L., and di Palma M., A., 2022. *robustbase: Basic Robust Statistics*. R package version 0.95-0, http://robustbase.r-forge.r-project.org/.

MERI. 2015. Measuring elevation change in Meadowlands marshes using Surface Elevation Tables (SETs) and Marker Horizons (MH). pp.1–14.

Neckles, H.A., Guntenspergen, G.R., Shriver, W.G., Danz, N.P., Wiest, W.A. Nagel, J.L., and Olker, J.H., 2013. Identification of metrics to monitor salt marsh integrity on National Wildlife Refuges in relation to conservation and management objectives. Final report to U.S. Fish and Wildlife Service, Northeast Region. USGS Patuxent Wildlife Research Center, Laurel, MD. 226 pp. https://www.pwrc.usgs.gov/prodabs/pubpdfs/7828_Neckles.pdf

Niering, W.A. and Warren, R.S., 1980. Vegetation Patterns and Processes in New England Salt Marshes, *BioScience*, *30*(5), pp. 301–307, https://doi.org/10.2307/1307853

R Core Team, 2017. R: A language and environment for statistical computing. R version 4.0.2. R Foundation for Statistical Computing, Vienna, Austria. https://www.R-project.org

Raposa, K.B. and Talley, D.M., 2012. "A meta-analysis of nekton responses to restoration of tide-restricted New England salt marshes." *Tidal Marsh Restoration: A Synthesis of Science and Management*. Washington, DC: Island Press: 97-118. Print.

Ray, G.L., 2007. Thin layer placement of dredged material on coastal wetlands: A review of the technical and scientific literature. ERDC/EL Technical Notes Collection (ERDC/EL TN-07-1), Vicksburg, MS: U.S. Army Engineer Research and Development Center.

Reimold, R.J., Hardisky, M.A., and Adams, P.C., 1978. The effects of smothering a *Spartina alterniflora* saltmarsh with dredged material. WES-TR-D-78-38. U.S. Army Waterways Experiment Station, Vicksburg, MS.

Tyler, A.C. and Zieman, J.C, 1999. "Patterns of development in the creekbank region of a barrier island Spartina alterniflora marsh." *Marine Ecology Progress Series,* 180: 161-177.

US EPA, 2019. Application of the sea-level affecting marshes model (SLAMM) to the lower Delaware Bay, with a focus on salt marsh habitat. pp. 1–74. US Environmental Protection Agency, Washington, DC.

Yohai, V.J., 1987. High Breakdown-Point and High Efficiency Robust Estimates for Regression. *The Annals of Statistics*, 15, pp.642-656. https://doi.org/10.1214/aos/1176350366