Evaluating coastal protection benefits of restored oyster reef designs

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Abstract

Wave attack is the driving force behind erosional processes causing marsh retreat in shallow coastal bays. Sea level rise and increased storminess are expected to heighten wave energy, escalating coastal erosion problems and intensifying vulnerability of coastal communities to storm damage. Thus, wave attenuation is a critical part of any coastal protection and resilience plan. Previous efforts to protect shorelines have largely involved bulkhead and seawall construction, both of which can be detrimental to nearshore habitats. Recent efforts have shifted towards the use of “living shorelines”, such as restored oyster reefs, as nature-based solutions for coastal protection that stabilize shorelines and enhance oyster populations and the ecosystem services they provide.

This study evaluated restored oyster reefs as a nature-based solution for attenuating waves, thus alleviating storm wave impacts, and stabilizing marsh edges in Wachapreague, Virginia. Different substrates, including oyster castles and new reef designs developed by Sandbar Oyster Company that consist of biodegradable hardscapes, were placed adjacent to a marsh island that affronts the Town of Wachapreague. Direct field measurements were collected before and after reef construction to quantify changes to the wave, morphologic and ecologic environments related to reef presence and development. Wave measurements were taken on either side of each reef to quantify changes in wave heights. Erosion pins were used to document in-situ change in marsh edge position at locations with and without fringing oyster reefs. Larval recruitment, oyster density and infaunal community structure data were also collected to evaluate the role of oyster reefs in habitat creation and enhancement.

Oyster reefs in this study successfully dissipated wind-wave energy, but only when water depths were near or below reef crest heights. In-situ measurements of reef-lined and un-lined

marshes showed reefs at this location significantly reduced rates of marsh edge erosion. This study demonstrates that marsh morphology and elevation are at least as important as the presence of fringing oyster reefs in reducing wave energy reaching marsh edges and stabilizing shorelines.

Both artificial reef designs in this study successfully fostered larval recruitment and oyster growth, with significantly higher oyster densities recorded on Sandbar substrate than oyster castles. The presence of constructed reefs had no significant effect on infaunal communities and sediment properties within the study period, highlighting the need for continued restoration monitoring. The current reef configurations at this location seem to be effective restoration designs for stabilizing the marsh edge and enhancing the oyster population. Results from this study supplement our understanding of the combination of restored oyster reefs and marshes as a nature-based solution to threats associated with storms and sea level rise.

Acknowledgements

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1. Introduction

As sea level rises and storms become more frequent, coastal communities are becoming more vulnerable, forcing landowners to initiate shoreline protection measures. Previous efforts to defend shorelines have largely involved bulkhead and seawall construction, both of which can be detrimental to nearshore habitats. There is new interest in the use of "living shorelines", such as the combination of restored oyster reefs and marshes, as nature-based solutions for coastal protection that stabilize shorelines and enhance oyster populations and the ecosystem services they provide.

1.1 The eastern oyster

While historically abundant, populations of the eastern oyster, *Crassostrea virginica,* have experienced immense decline over the last century. In the Chesapeake Bay, this decline was mainly due to overharvesting and subsequent habitat loss, but also disease vulnerability and water quality (Rothschild et al., 1994). Within the coastal bays along the Atlantic Ocean side of the Delmarva Peninsula, Virginia, the decline is not well documented; however it is thought to follow similar patterns as those in the Bay. Population reduction was most likely caused by a combination of disease presence and overexploitation at this location (Ross and Luckenbach, 2009).

More recently, efforts have been made to restore oyster populations and the ecological, water quality, fisheries production and coastal protection benefits they provide. The native *Crassostrea virginica* is frequently referred to as an "ecosystem engineer" because it creates three-dimensional complex reef structures within the intertidal regions of shallow coastal bays and estuaries that influence ecological communities and physical environments (Jones et al., 1994). Artificial reefs can affect infauna abundance and community structure through alterations of local biologic and/or physical environments. Reefs provide a habitat protected from waves that may enhance the abundance of some infauna taxa and depress the abundance of others, perhaps due to foraging by predators (Ambrose and Anderson, 1990; Hogan and Reidenbach, 2021). Oyster reefs provide the foundation for tropically diverse benthic communities, thus increasing biodiversity, preserving habitat connectivity and maintaining ecosystem integrity (van der Zee et al., 2012; van der Zee et al., 2015; Breitburg et al., 2000; Lenihan and Peterson, 1998; Eggleston et al., 1999; Coen et al., 2007; Scyphers et al., 2011).

Oysters improve water quality through uptake of suspended particles, phytoplankton and dissolved organic matter by filtration across their gills (Jorgensen, 1990). This suspension feeding aids in nutrient cycling, where organic matter is taken in by oysters and inorganic nutrients rejected by the animal are expelled into the water column as pseudo feces (MD Sea Grant "Particulate Matters", 2021). Oyster pseudo feces, nutrient remineralization and burial can lead to increased organic matter content in sediments surrounding reefs, however this can vary with ecosystem physical-biological interactions (Southwell et al., 2017). Some studies have shown that oysters retain larger particles (> 2 µm) through filtration, resulting in a greater presence of fine grains in benthic sediments (Jorgensen, 1990; Southwell et al., 2017; Colden at al., 2016). Alternatively, others have shown that sediment becomes coarser in the presence of reefs, perhaps due to wave protection and benthic sediment stability decreasing erosion of larger sediments (Taube, 2013; Ambrose and Anderson, 1990). Oyster reefs are capable of influencing ecological communities through alterations of biological and/or physical environments, however their effects are dependent on highly complex, location-specific ecosystem interactions.

Not only do oyster reefs serve important roles in local ecology, but they also have the ability to buffer wave energy and stabilize shorelines. The use of fringing oyster reefs for shoreline protection has been investigated along Virginia's Eastern Shore (Wiberg et al., 2019; Taube 2013; Ferguson 2018; Hogan et al., 2021) and in the Gulf of Mexico and Mid-Atlantic (Piazza et al., 2005; Meyer et al., 1997, Stricklin et al., 2010) among other locations. These studies show oyster reefs as protective breakwaters that reduce wave impacts (Wiberg et al., 2019; Taube, 2013) and stabilize shorelines allowing for accretion along marsh edges (Piazza et al., 2005; Meyer et al., 1997; Stricklin et al., 2010, Hogan et al., 2021). Documentation generally suggests that more solid vertical reef structures are required for erosion control in mid-energy conditions (Scyphers et al., 2011), while shell piles work best in low energy areas where they are less likely to be buried or washed away (Piazza et al., 2005, Stricklin et al., 2010). Hogan and Reidenbach (2021) found greater attenuation over higher elevation reef structures (0% vs 21%, low vs high elevation) and increased attenuation as reefs matured (13% vs 21%, year 1 vs year 3).

1.2 Oyster restoration efforts

Restoration projects aimed at increasing habitat and improving the aquaculture industry have proven successful and oyster surveys conducted during 2007-2008 in Virginia's coastal bays suggest a self-sustaining population with the potential for significant expansion (Ross and Luckenbach, 2009). While fishing regulations have improved and oysters have developed greater resistance to disease, recruitment and population increase is limited by the presence of suitable substrate. It has been shown that successful oyster larval recruitment, survival and growth relies on hard substrate for attachment (Whitman and Reidenbach, 2012) and that larvae prefer to settle on existing oyster reefs (Butman, 1987). Therefore, a main objective of restoration is to create hard substrate that can act as a suitable attachment location for oyster larvae to anchor and grow to maturity.

Additionally, larval recruitment and survivability are affected by local hydrodynamics and reef elevation. High flow velocities transport and increase larval supply and substrate encounters and reduce mortality from burial caused by sedimentation (Lenihan, 1999). Larvae use turbulent mixing as an indicator of suitable substrate presence, but after landing, low shear stresses within the interstitial areas between oysters or other roughness elements allow them to remain anchored (Whitman and Reidenbach, 2012; Reidenbach et al., 2009; Crimaldi et al., 2002). Therefore, while fast moving water acts to transport larvae, they often settle and grow in a microhabitat where lower stresses decrease the risk of displacement after attachment (Hubbard and Reidenbach, 2015; Whitman and Reidenbach, 2012; Reidenbach et al., 2009). Suitable habitat is also largely dictated by reef elevation and oyster densities and shell heights tend to be greater on higher elevation than lower elevation reef crests (Hogan and Reidenbach, 2021; Schulte, 2009).

Traditionally, oyster restoration projects have focused on alternative reef materials such as piles of oyster and other bivalve shells and oyster castles. Shell piles and oyster castles provide substrates that mimic some of the aspects of natural reefs and have been shown to promote larval recruitment, oyster growth and survival (Theuerkauf et al., 2015). Alternatively, a new substrate developed by Sandbar Oyster Company consisting of biodegradable hardscapes, may provide a framework even more similar to that of a natural reef with increased surface area for attachment and growth (Sandbar Oyster Company, 2021). While early restoration efforts focused on reviving oyster populations, recent efforts have shifted to also focus on the coastal protection benefits, including that of marsh-bay boundaries, provided by oyster reefs.

1.3 Marshes as natural buffers

Saltmarshes are located in intertidal areas where there is ample sediment supply with favorable plant-growing conditions and salt-tolerant marsh grasses. These systems are regularly inundated by tides and contain vegetation that, once established, traps sediment causing accumulation that forms and stabilizes the marsh platform (Zedler et al., 2008). Similar to oyster reefs, marshes play vital ecologic and economic roles – they serve as habitat and breeding grounds for many plants, migratory bird species and other animals, sequester carbon, filter nutrients, trap sediment thereby reducing turbidity and aid in fisheries production (Boorman, 1999; Zedler et al., 2008; Boesch and Turner, 1984). Saltmarshes are also particularly critical to shoreline protection as they are often a last defense to wind waves.

Saltmarshes are naturally dynamic systems that are constantly evolving due to surface, interior and edge erosion and deposition and biotic processes. Erosion due to anthropogenic and natural processes such as waves, land subsidence and sea level rise, make marshes vulnerable to losses, which have been documented worldwide (Fagherazzi et al. 2020; Phillips, 1986; Gedan et al., 2009). Wave attack is the driving force behind erosional processes causing marsh-edge retreat in shallow coastal bays (Mcloughlin et al., 2015; Fagherazzi and Wiberg, 2009) and wave energy is positively correlated with water depth, wind speed and fetch (Mariotti et al., 2010). Thus, relative sea level rise (RSLR) and more frequent storms are forecasted to increase wave energy and escalate marsh erosion problems (Mariotti et al., 2010; Mcloughlin et al., 2015). Many coastal communities that rely on the natural buffering capacity that marshes provide from winds and waves are becoming more exposed and vulnerable as these systems begin to erode and disappear globally (Gedan et al., 2009), creating a need for sustainable adaptation strategies (Temmerman et al., 2013).

1.4 Approaches to shoreline protection

One traditional approach to mitigating shoreline erosion, including marsh-bay edges, is the use of hard shoreline stabilization such as seawalls and bulkheads (grey matter) built parallel to the shore, usually constructed of concrete, wood or steel (Gittman et al., 2016). Despite evidence of the degradation of coastal ecosystems resulting from hard shoreline stabilization techniques, this type of infrastructure is common along both sandy and marshy coasts in Virginia (Duhring et al., 2006) because of its effectiveness at reflecting waves away from the shore (Plant and Griggs, 1992). Consequently, these structures may act to intensify the area's vulnerability and undermine its resilience by interrupting natural physical, biological and geological processes (Gittman et al., 2016).

Grey matter is widely used for shoreline stabilization on Virginia's coast in part because of the perceived lower cost and lack of awareness about the benefits of natural infrastructure for shoreline stabilization (NFWF, 2019). More recent efforts have shifted towards the use of natural habitats, such as the combination of oyster reefs and saltmarshes, for coastal protection. These "living shorelines" minimize the use of grey infrastructure and encourage the use of natural structures capable of buffering and absorbing, rather than reflecting, wave energy. Living shorelines have proven effective at mitigating coastal erosion while also sustaining ecological communities (Currin et al., 2010).

Oyster reefs have been found to protect marsh edges under certain physical conditions. Waves generated by smaller, more frequent moderate storms are more likely to affect marsh boundaries than waves generated by large storms accompanied by storm surges (Leonardi et al 2016), and it is these intermediate waves that are most impacted by intertidal oyster reefs (Wiberg et al., 2019; Taube, 2013; Piazza et al., 2005). Wiberg et al. (2019) found that oyster reefs reduced wave heights by 30-50% for shallow water depths (reef crest heights near water surface) and 0-20% for deeper water waves (reef crest heights > 0.25 m below water surface). This study concluded that marshes with edge elevations close to mean sea level (MSL) are more likely to benefit from reductions in wave energy associated with oyster reefs than marshes with high edge scarps (Wiberg et al., 2019).

Fringing artificial oyster reefs are capable of reducing wave energy and erosion, but their effectiveness depends on hydrodynamic conditions, marsh elevation, water depth, and reef elevation. Marsh grasses have been shown to stabilize sediments and attenuate waves, even in extreme, deep water conditions where oyster reefs are not as effective (Moller et al., 2014; Ferguson, 2018). As marsh area accretes horizontally and creates more suitable habitat for marsh grass growth, attenuation increases (Zedler et al., 2008; Moller et al., 2014). The combination of constructed oyster reefs with vegetated marsh area can be an effective and sustainable long-term shoreline stabilization technique in which the reef helps to stabilize the marsh edge while marsh grasses attenuate wave energy that passes over the reef (Ferguson, 2018).

1.5 Nature-based solutions for coastal protection

The use of living shorelines, such as oyster reefs and marshes, for coastal protection is an example of a nature-based solution (NBS). NBS are defined by the International Union for the Conservation of Nature (IUCN) as "actions to protect, sustainably manage, and restore natural or modified ecosystems, that address societal challenges effectively and adaptively, simultaneously providing human well-being and biodiversity benefits" (IUCN, 2021). By providing both shoreline accretion and ecosystem benefits, oyster reef restoration may offer a nature-based solution to the protection of marshes, and therefore coastlines housing vulnerable communities, in the face of increased storm impacts caused by climate change and sea level rise. To date, there are no studies of the coastal protection benefits or ecological effects of oyster reefs constructed using the new Sandbar Oyster Company substrates alone or in combination with more traditional restored reefs, such as shell piles or oyster castles.

2. Study Area

2.1 Virginia Coast Reserve

The Virginia Coast Reserve (VCR) is a 100 km barrier-lagoon-marsh system on the Atlantic Ocean side of the Delmarva Peninsula, USA (Figure 1). The VCR, one of the nation's last remaining stretches of undeveloped coastal wilderness protected by the Nature Conservancy (TNC) and its partners, contains a string of fourteen barrier islands that protect diverse landscapes including mainland saltmarshes with tidal creeks, mudflats, coastal bays, oyster reefs and seagrass meadows. Not only do these natural, continuous coastal systems play important ecological roles, but they also help to protect coastal communities on the Eastern Shore of Virginia. As a largely natural but threatened coastal region, the VCR offers an ideal setting in which to study and demonstrate the capacity of nature-based solutions to mitigate risks from the combined impacts of rising sea levels and storms, which exacerbate erosion of coastal habitats (NFWF, 2019).

The Town of Wachapreague (approximately 37° 36' 13.79" N, -75° 41' 15.59" W), located in southern Accomack County, Virginia in the northern portion of the VCR, is a low-lying waterfront community protected by barrier islands backed by coastal bays. Wachapreague's economy relies heavily on commercial fishing and research at William & Mary's Virginia Institute of Marine Science (VIMS) lab, which is built on the mainland shoreline. The VIMS lab is host to a National Oceanic and Atmospheric Administration (NOAA) tide a meteorological station that has monitored water levels, winds and temperatures since 1978.

The shallow bays of the VCR undergo a semidiurnal tidal range of roughly 1.2 m within the microtidal range (less than 2 m). The rate of relative sea level rise (RSLR) in Wachapreague is 5.52 +/- 0.66 mm $\cdot$ yr^{-1} over the last 42 years (NOAA Sea Level Trends, 2021). Dominant wind directions in the study area are from the NNE and SSW with the strongest coming from the N and most frequent from the SSW (Wiberg et al., 2019). Vegetation in the low, intertidal salt marshes of this area is dominated by *Spartina alterniflora*, commonly known as cordgrass (Brinson et al., 1995) with an average height of 40 cm for the short-form and 100 cm for the intermediate-form (Blum, 1993).

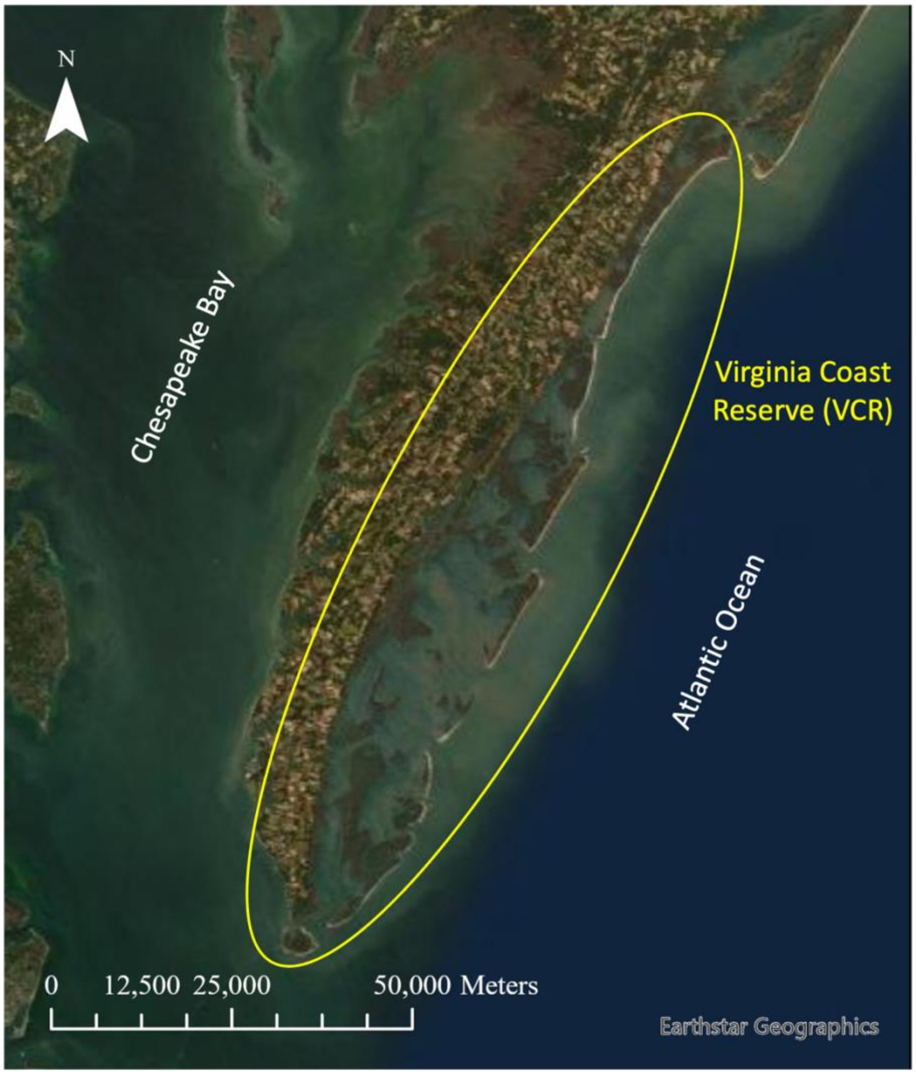

Figure 1. Map of the Virginia Coast Reserve (VCR).

2.2 Restoration site selection and description

Aerial photographs taken over a period of 17 years have documented significant reduction in the area of a saltmarsh just seaward of Wachapreague (Figure 2). This marsh island acts as a first line of defense by buffering wind waves reaching the town by ~90% (Figure 3). As a result, this saltmarsh island was selected as a site for oyster reef restoration. This location is also where the United States Army Corps of Engineers (USACE) is currently depositing thin layers of dredged material to increase the marsh's elevation to aid in its ability to keep up with sea level rise, thus increasing the marsh's ability to protect against storms (NFWF, 2019).

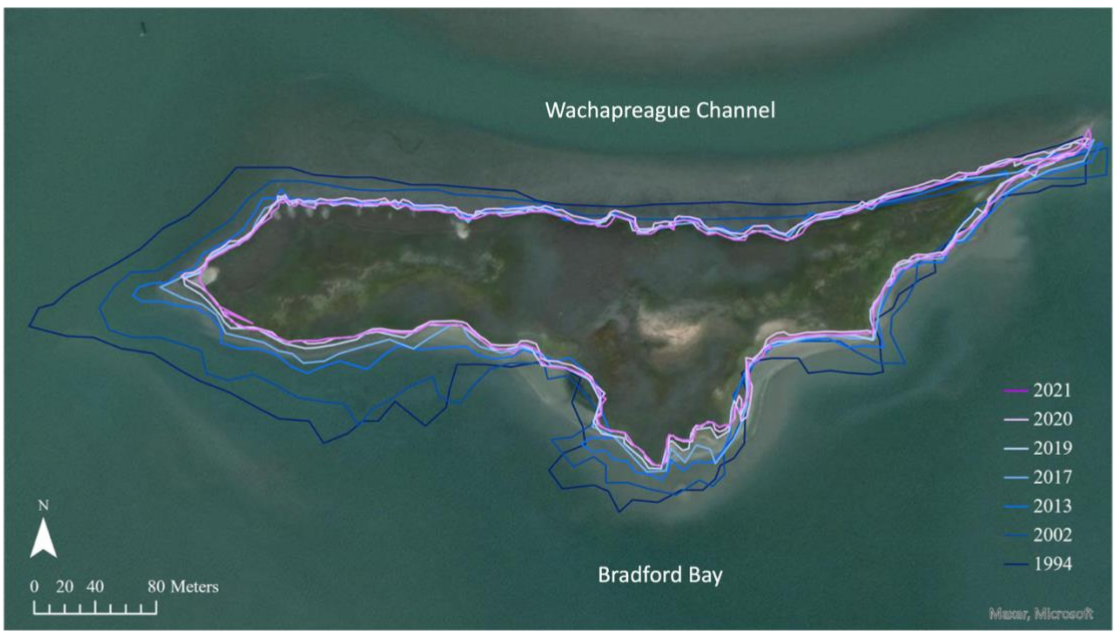

Figure 2. Change in marsh edge position and area from 1994 to 2021.

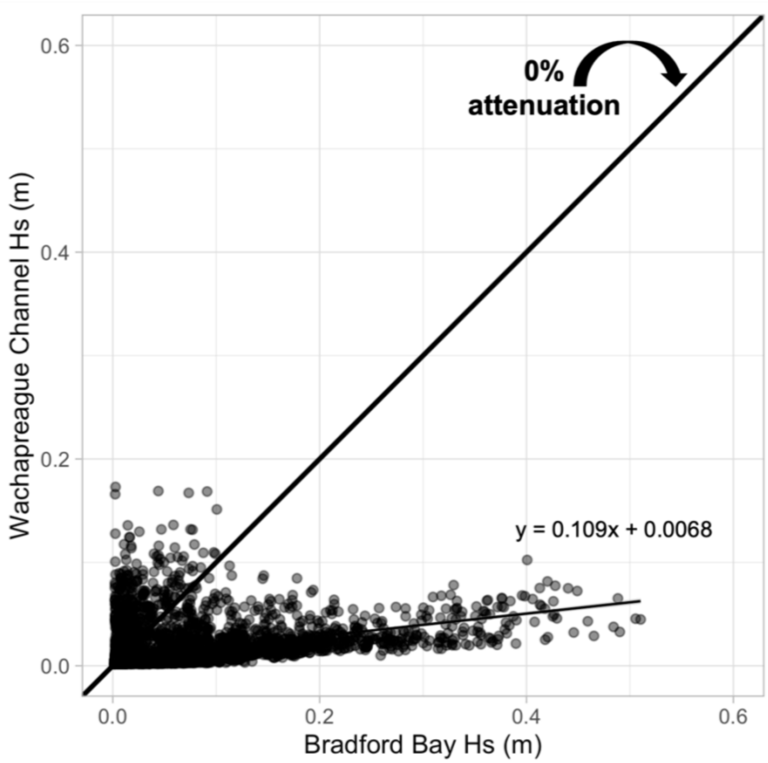

Figure 3. Regression analysis of significant wave heights (*Hs*) at Bradford Bay, the more exposed side of the marsh, and Wachapreague Channel, the more protected side. A slope of 0.109 suggests a ~90% reduction in wave heights from Bradford Bay to Wachapreague Channel.

For the reef restoration project, the marsh shoreline was divided into 9 sections that were treated with different oyster substrate designs, including oyster castles and new substrates developed by Sandbar Oyster Company, depending on certain geomorphologic characteristics discussed below. TNC managed oyster substrate construction and deployment while assessment and monitoring of changes to the hydrodynamic, morphologic and ecologic environments at two of the marsh sections, 4 and 7 (Figure 4), was completed by UVA through a National Fish and Wildlife Foundation (NFWF) grant.

Figure 4. Project location in relation to the Town of Wachapreague.

Oyster castles

The oyster castles in this study are notched concrete blocks developed by Allied Concrete, Inc. in Charlottesville, VA (Figure 5). The castles are designed to latch together to create vertical structures that provide both resistance against wave energy and substrate for attachment. These types of oyster reefs have been previously used across the VCR. Oyster castles are ideal for flat, hard bottom areas and for this project, Section 7 contains two rows of oyster castles (Figure 6).

Sandbar substrate

Sandbar Oyster Company has developed several alternative oyster substrate designs that were created for conditions in which oyster castles are less successful. These designs consist of biodegradable hardscapes composed of plant fiber cloths with mineral-based binders that are wet with concrete and formed into different shapes and sizes (Figure 5). Section 4 in this project contains a combination of oyster castles and Sandbar substrates that are ideal for areas with a firm, more level bottom offshore and uneven, muddy bottom at the marsh scarp, respectively (Figure 6). While Sandbar Oyster Company substrates have previously been used in locations in North Carolina (Sandbar Oyster Company, 2021) and the Chesapeake Bay (Chesapeake Bay Foundation, 2021), this is the first use of them in the VCR.

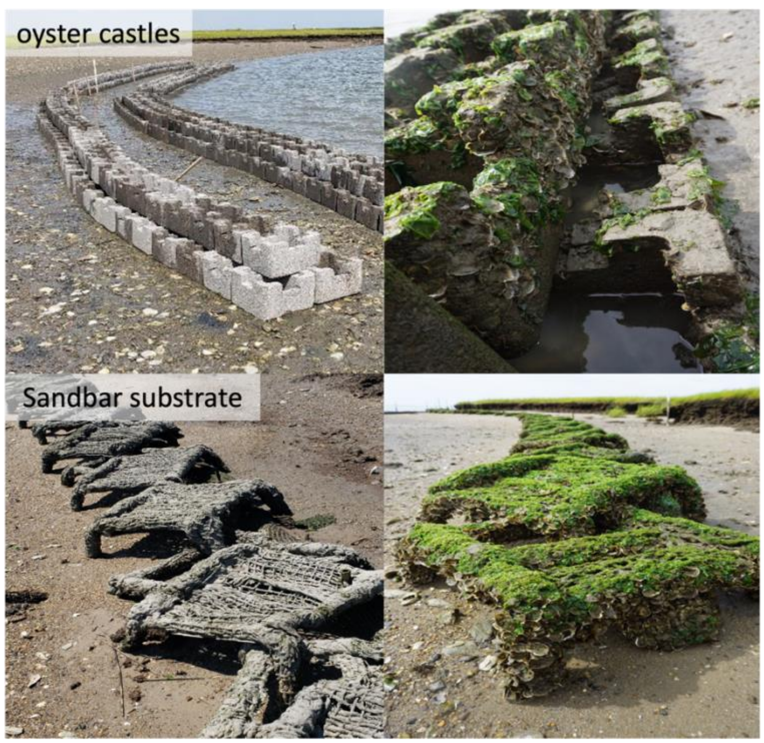

Figure 5. The two types of oyster substrate, oyster castles and Sandbar substrate, used in this study just after deployment (left) and several months after deployment with some oyster strike (right).

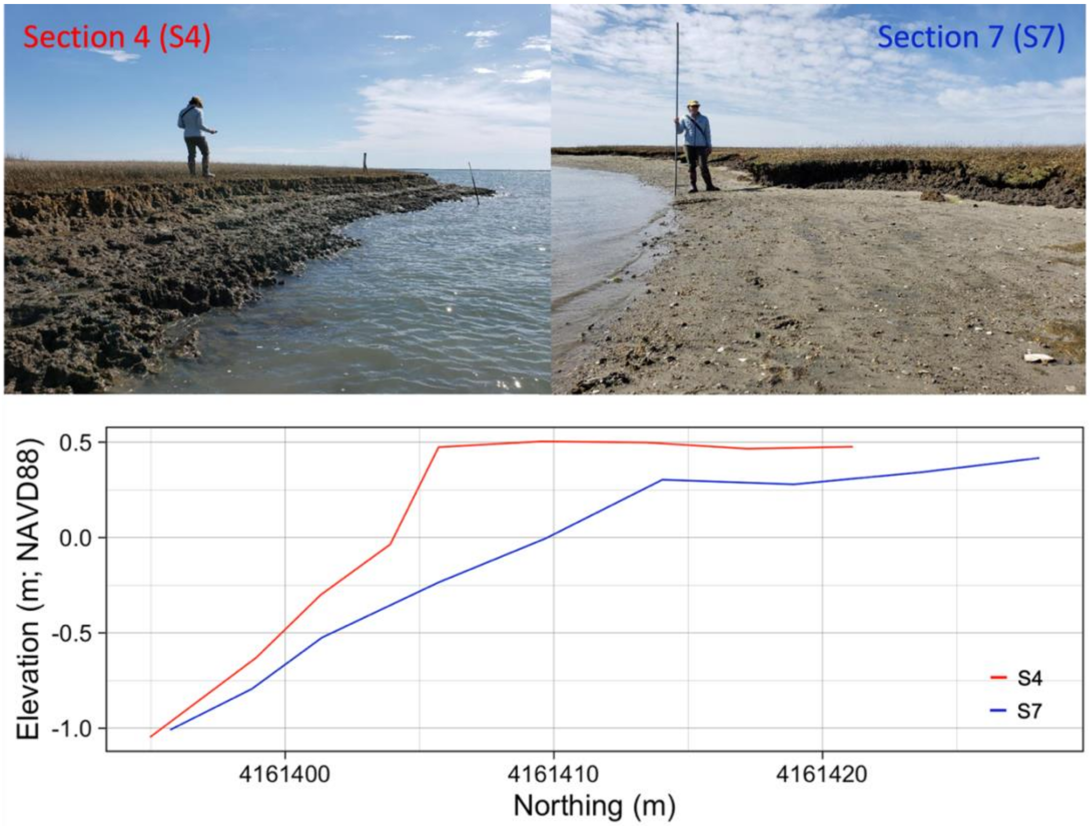

Figure 6. Cross sectional profiles of Sections 4 and 7 of the shoreline.

3. Objectives & Approach

Previous studies suggest that restored oyster reefs aid in nature-based coastal protection through the alteration of local hydrodynamics, marsh morphology and ecological community structure. The main objective of this study is to evaluate coastal protection benefits of restored oyster reef designs, including oyster castles and the novel Sandbar substrate, through direct measurements of physical, morphologic and ecologic changes associated with reefs constructed adjacent to a marsh affronting Wachapreague, VA. The primary aims are to:

1. Study the effectiveness of restored oyster reef designs at attenuating waves, increasing marsh area and altering edge morphology.

2. Document the role of restored oyster reef designs in habitat creation and enhancement.

The following research questions were asked to address the first primary aim: (1) To what degree are artificial oyster reef designs capable of attenuating waves?, (2) Does oyster reef design have an impact on marsh edge erosion and sedimentation?

These questions were answered through hydrodynamic and morphological change measurements before and after reef construction. Wave attenuation was quantified as the change in wave heights over the reefs. Change in marsh edge position was tracked in-situ using erosion pins and remotely through aerial photography analysis. Marsh sedimentation was measured by placing sediment accumulation plates behind the reefs. These data were collected before and after reef construction to quantify changes to the physical and morphologic environments associated with reef placement and development through time. These measurements help determine the role that fringing artificial oyster reefs play in the accretion of a marsh affronting Wachapreague's coast.

The following research questions were asked to address the second primary aim: (1) How does substrate affect oyster recruitment, abundance and shell height?, (2) How do sediment, infauna and marsh grass properties differ post reef construction?

These questions were answered through field study of the local ecology associated with the oyster reefs. Recruitment success and population stabilization was determined using larval settlement, recruit and adult density and shell height measurements. To understand how artificial oyster reefs affect the intertidal community at this location, sediment and infaunal community structure data were collected before and after reef construction. These measurements aid in our understanding of the role that oysters can play in habitat creation and enhancement at this location.

4. Methods

As part of this project, two sections of the shoreline, 4 and 7 (S4 and S7), were monitored several times throughout two years. S4 contains a row of oyster castles and row of Sandbar substrate while S7 contains two rows of oyster castles (Figure 4). Site monitoring consisted of hydrodynamic, sediment accumulation, morphologic and ecologic measurements. Larval recruitment and oyster density measurements were conducted annually. Pre-construction monitoring took place in fall 2020 while post-construction monitoring took place summer 2021 through summer 2022 (Figure 7).

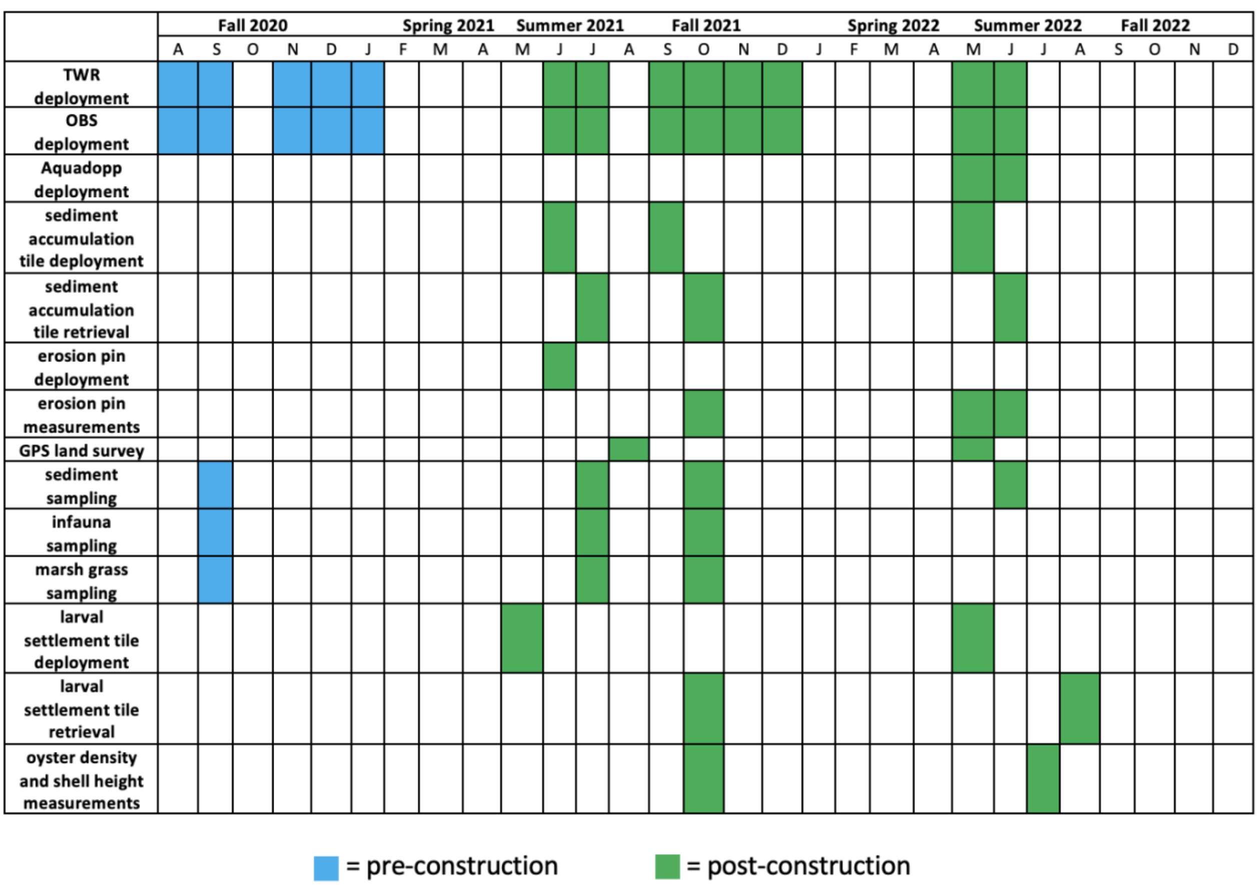

Figure 7. Timeline of deployments and measurements taken before and after reef construction.

4.1 Hydrodynamics

Water levels and wave conditions were measured at Sections 4 and 7 of the shoreline several times a year along approximately 20 m transects that extend across both sides (bayward and landward) of the constructed reef locations (Wiberg et al., 2019; Taube, 2013, Hogan and Reidenbach, 2021) (Table 1). Measurement locations were named Sections 4 and 7 Nearshore (landward) and Offshore (bayward) and will be referred to as S4N, S7N, S4O and S7O, respectively. An additional sensor was placed between the rows of oyster castles and Sandbar substrate at Section 4 in July and November 2021 and will be referred to as S4M (Figure 8).

Table 1. Deployment schedule with sensor type and locations.

Start-End Date	Pre/post reef construction	Instruments Deployed	Sensor Locations
Aug 31-Sep 27, 2020	pre	TWR	S4N, S4O, S7N, S7O
Nov 27, 2020-Jan 15, 2021	pre	TWR	WC, BB
Jun 25-Jul 26, 2021	post	TWR	S4M, S4O, S7N, S7O
Nov 11-Dec 11, 2021	post	TWR	S4N, S4M, S4O
May 19-Jun 13, 2022	post	TWR, OBS, Aquadopp	S7N, S7O, S2N, S2O

Richard Branker Research (RBR) Tide and Wave Recorders (TWR) (RBR Ltd., Ontario, Canada), or wave gauges, were deployed for a minimum of three weeks, a full spring-neap cycle, several times a year to capture a range of both typical and storm conditions. The simultaneous deployment of wave gauges on either side of the reefs allowed for the quantification of attenuation as waves propagated towards the marsh. Wave gauges were attached to metal grates, secured flush with the bay bottom using sand stakes and made visible from the surface with a buoy (Figure 9). Wave conditions were measured over a sampling period of 10 minutes using a sample speed of 4Hz with a burst height (samples) of 512.

Following field collection, atmospheric pressure data recorded at the NOAA station (Station ID: 8631044) in Wachapreague, VA were used to correct pressure measured by the wave gauges, which was then converted to water depth (NOAA Tides and Currents, 2022). Comparison of water depth measurements from NOAA's Wachapreague station and the RBR wave gauges showed excellent agreement (Figure 10). Depths were also calculated relative to mean water level (MWL), or the average water depths recorded by offshore sensors, for each section.

Significant wave height (Hs) and mean wave period ($Tavg$) were calculated following the methods in Wiberg and Sherwood (2008). The nearshore wave gauges were often exposed during low tides. Average wave period could not be calculated when the wave gauge was out of the water for a significant portion of the sampling period (shallow water conditions); during these times, wave height and period were assigned NA and were not included in subsequent calculations. Data from the outer (bayward), inner (landward) and middle (between oyster castles and Sandbar substrate) wave gauges were separated into shallow ($\leq 0.05\ m$ above nearshore reef crest), intermediate ($> 0.5\ m$ above nearshore reef crest to water depth at marsh platform) and deep (≥ water depth of marsh platform) water (Wiberg et al., 2019; Hogan and Reidenbach, 2021; Ferguson, 2018) for analysis.

A RBR Optical Backscatter Sensor (OBS) (RBR Ltd., Ontario, Canada), deployed simultaneously with a wave gauge, was used to measure turbidity at S7O in June 2022. An OBS was also deployed at S4, however the data were not usable. The OBS sensor used in this study was calibrated in-lab during the summer of 2022 using sediment from S7O to allow turbidity readings to be converted to suspended sediment concentrations.

A Nortek© High-resolution Acoustic Doppler Profiler (Aquadopp) was used to characterize current velocities throughout the water column and provide information about general circulation patterns near the reefs. This instrument was deployed simultaneously with a wave gauge in Spring 2022 at S7O and recorded velocity every 30 mins at 1 Hz for several weeks (Taube, 2013; Wiberg et al., 2019).

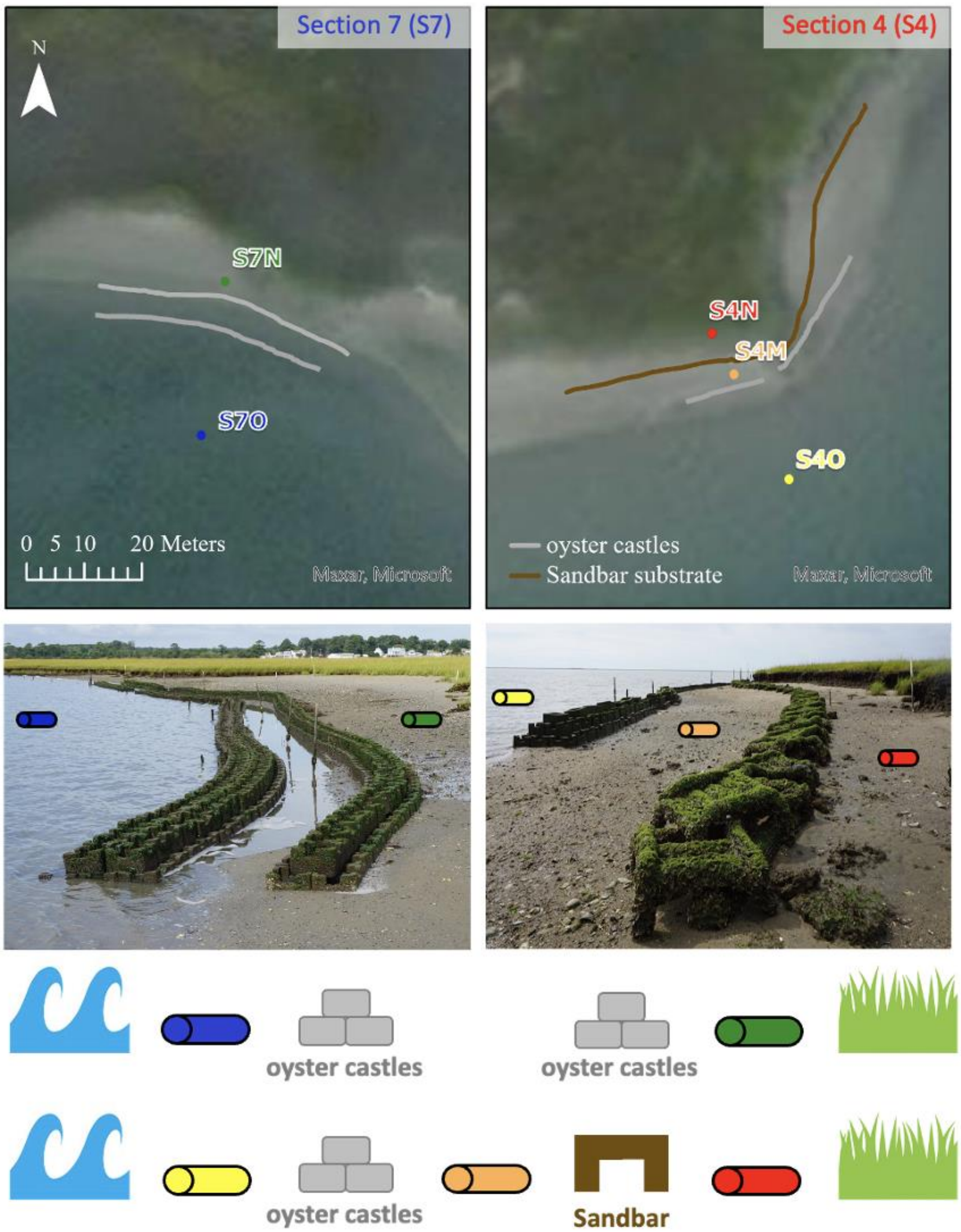

Figure 8. Wave gauge and sediment sensor deployment locations at Sections 7 and 4.

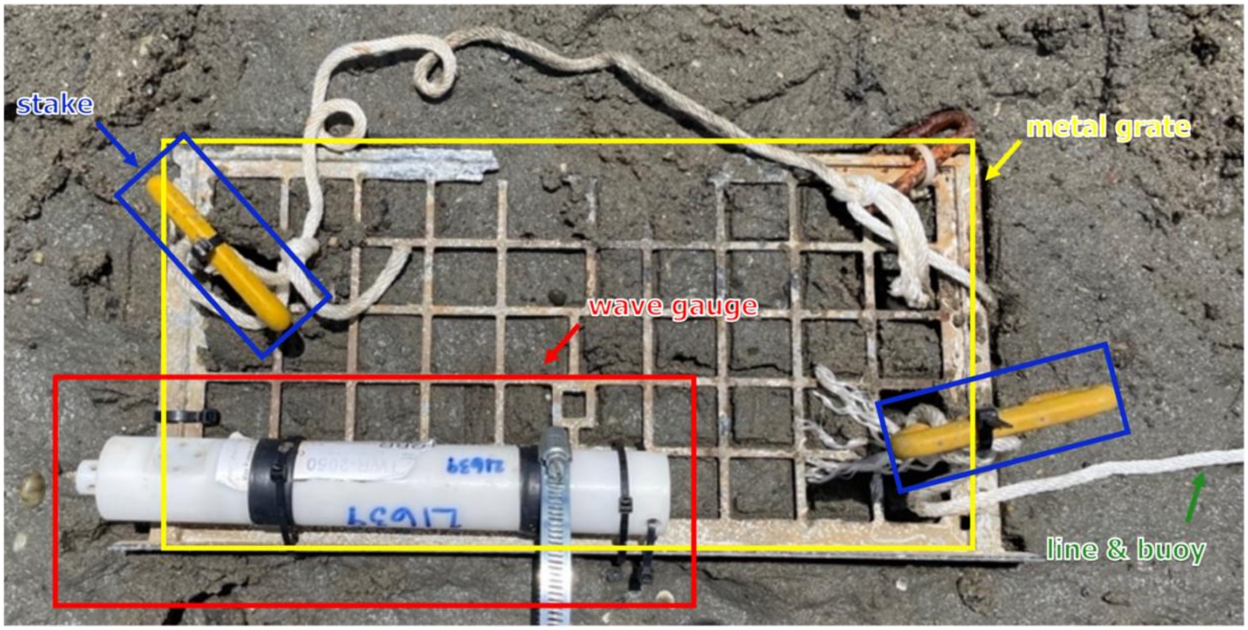

Figure 9. Wave gauge secured to a metal grate with zip ties, staked into the ground and made visible from the water surface with a buoy.

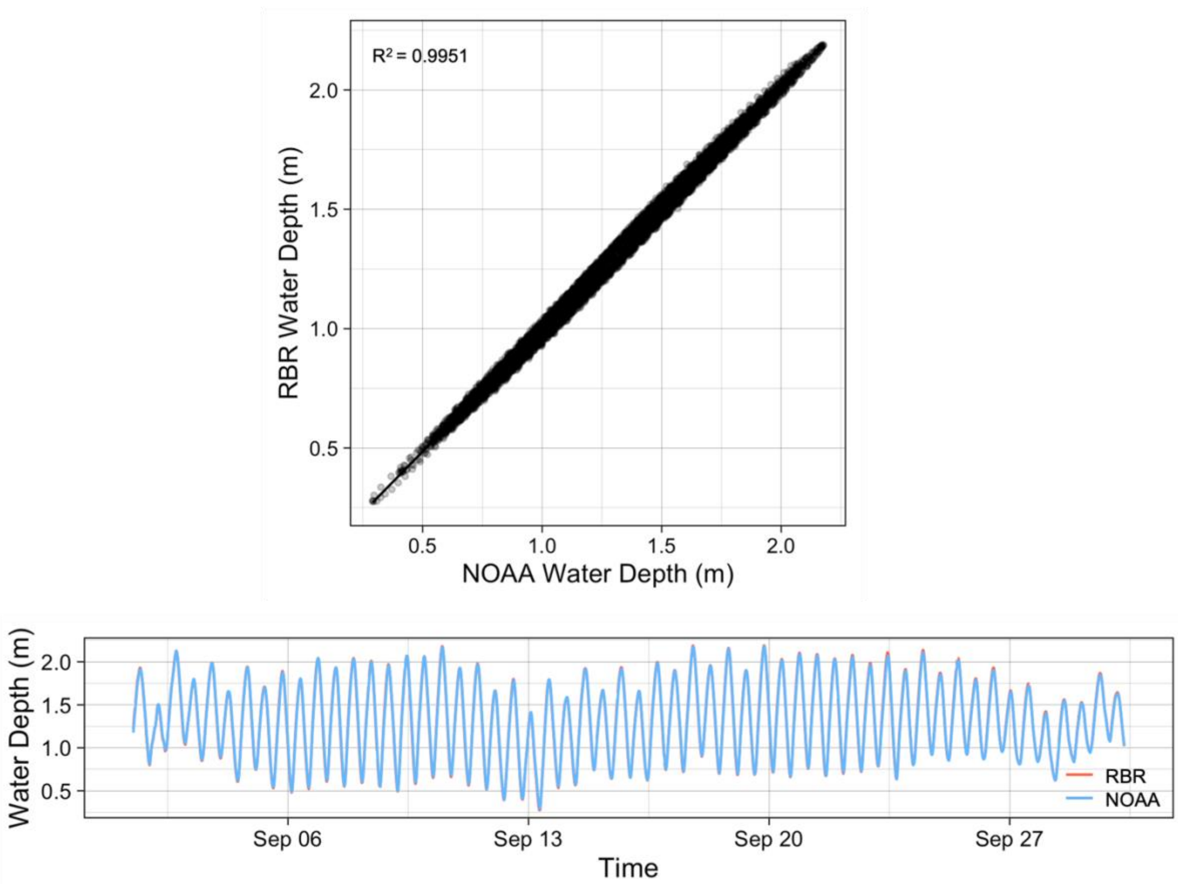

Figure 10. NOAA Wachapreague station water depth measurements compared to RBR water depth measurements from September 2021 from S4O.

4.2 Sediment accumulation

Sediment accumulation was measured coinciding with wave measurements. Sediment accumulation plates (square ceramic tiles) were deployed directly onto the marsh platform. A shovel was used to dig out sediment so that the plates lay flush with the surface. Three tiles were placed approximately 5 m apart along two transects from the marsh edge to the interior at Sections 4 and 7 (12 tiles total) (Figure 11). Plates with accumulated sediment were covered with plastic, transported back to the lab and oven dried at 60°C to give total accumulated mass (Kastler and Wiberg, 1996). One-way ANOVA and Tukey multiple comparisons tests were used to determine difference in sediment accumulation between sections (4 and 7) and location on the marsh (edge, middle and interior).

Figure 11. Locations of sediment accumulation tile deployment.

4.3 Marsh edge morphology and elevation

Change in marsh morphology over time was tracked remotely through aerial photograph analysis and in-situ using erosion pins and land surveys. For aerial photograph analysis, photos were chosen based on availability, time intervals and image quality. The images were given spatial context through the georectification tool in ArcGIS Pro 2.6 using landmarks with a x and y coordinate, such as the edge of a building or road intersection, as ground control points. A new feature class was created in ArcGIS Pro 2.6 to trace and digitize shorelines (Figure 2). The vegetation line was used as a shoreline indicator because of its visibility and independence of tide (Taube, 2013).

In the field, PVC marker poles were inserted into the sediment along the marsh shoreline at several meter intervals during the summer of 2021 (Figure 12). These markers were used as a visual key for any short-term change. Distances between the PVC poles and the new shoreline were measured to quantify edge change at each pole site. In-situ and remotely-sensed shoreline changes were compared.

Land surveying was conducted on the marsh island in summer 2021 to obtain precise marsh elevations, edge profiles and reef crest heights. Sixty-two survey points were taken along the perimeter of the marsh island to record edge position and elevation. Additionally, two transects perpendicular to the shorelines at S4 and S7 were used to create marsh elevation profiles from the marsh interior to just offshore of the reefs (Figure 13). Three survey points were also collected on each row of oyster reefs at S4 and S7 to obtain average reef crest elevations. Points were collected via using either a Trimble© R8 GNSS system (Trimble, Sunnyvale, CA) or an Emlid RTK GNSS/GPS receiver in summer 2021 and spring 2022. Survey locations collected using the Trimble system were corrected using NOAA's Online Positioning

Service (OPUS) and processed in Trimble Geomatics Office (TGO) while those collected with the Emlid were corrected and returned immediately using the ReachView 3 mobile app.

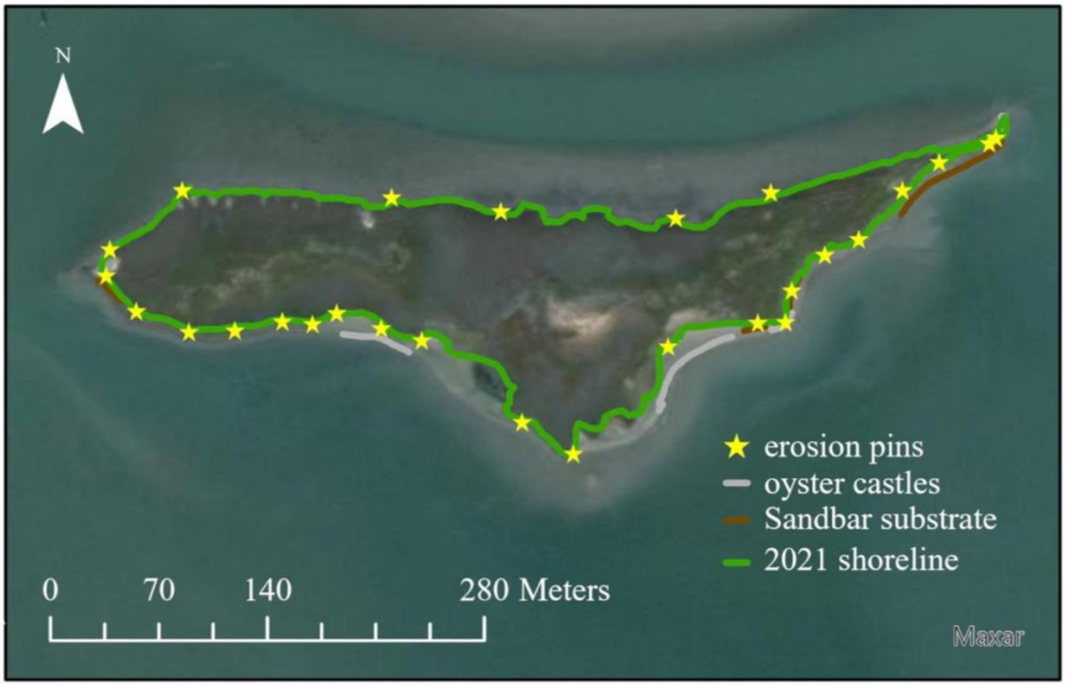

Figure 12. Locations of erosion pins (PVC poles) deployed at the marsh edge in June 2021. Erosion pins on the backside (northern edge) of the marsh island were not used in retreat analysis.

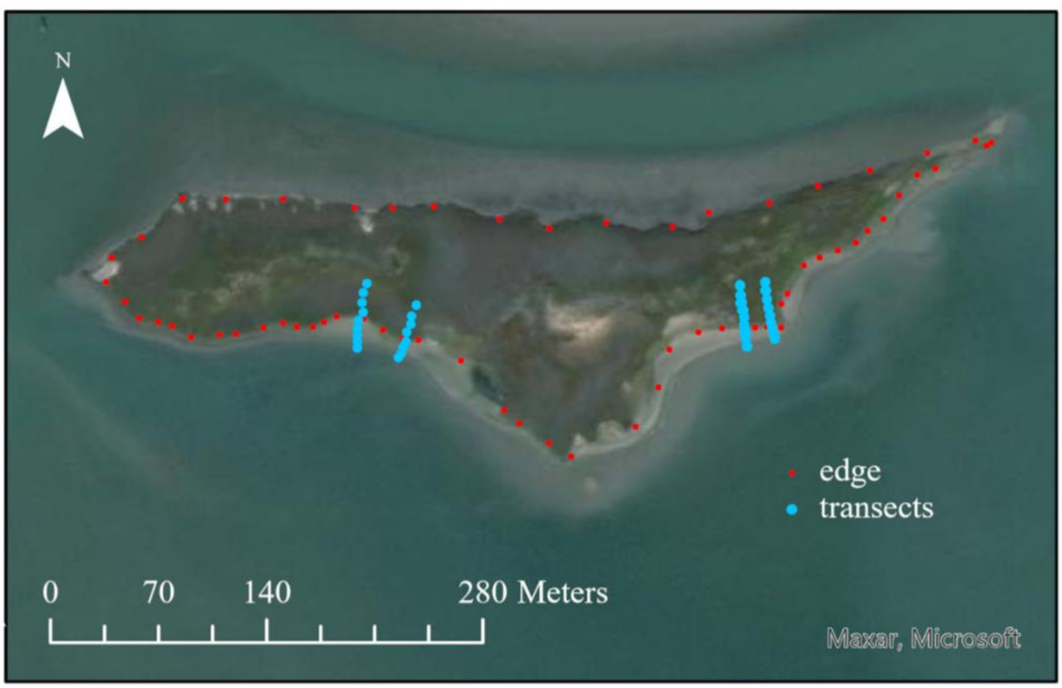

Figure 13. Locations of edge and transect survey points taken in August 2021.

4.4 Sediment properties

Samples were collected for sediment property analysis several times a year, coinciding with hydrodynamic measurements. During each collection period, eight sediment samples, two from each S4N, S7N, S4O and S7O were gathered for sediment property analysis, including grain size and organic matter content, using 5 cm adapted syringe corers. Four samples were analyzed for grain size and four for organic matter content at each site, two at S4N and S7N and two at S4O and S7O (Figure 14).

In the lab, samples used for organic matter content analysis were placed in a drying oven to determine sample dry weight (g) and then placed in a furnace to determine sample ash weight (g). Organic matter content of each sediment sample was determined by the difference between sample dry and ash weights (Taube, 2013). One-way ANOVA and Tukey multiple comparisons tests were used to determine difference in percent sediment organic matter content between dates.

Sediment grain size, an important factor in sediment resuspension, was characterized using a Bettersizer ST Laser Particle Size Analyzer (PSA). Prior to PSA use, sediment samples were wet sieved using 1 mm mesh and placed in glass jars to settle. Excess water was siphoned off and samples was placed in centrifuge tubes with 10 mL of water. To oxidize organic matter, samples were treated with hydrogen peroxide in a 80°C shaking water bath, sodium acetate and then centrifuged to remove carbonates and soluble salts. A 5% sodium hexametaphosphate solution was added to each sample as a dispersant to allow for deflocculation. PSA results gave percent composition of each size class and average median (d_{50}) particle size of three sub-samples for each sample. Sub-sample averages for each sample (2 samples per location) were averaged for each location (S4N, S4O, S7N and S7O).

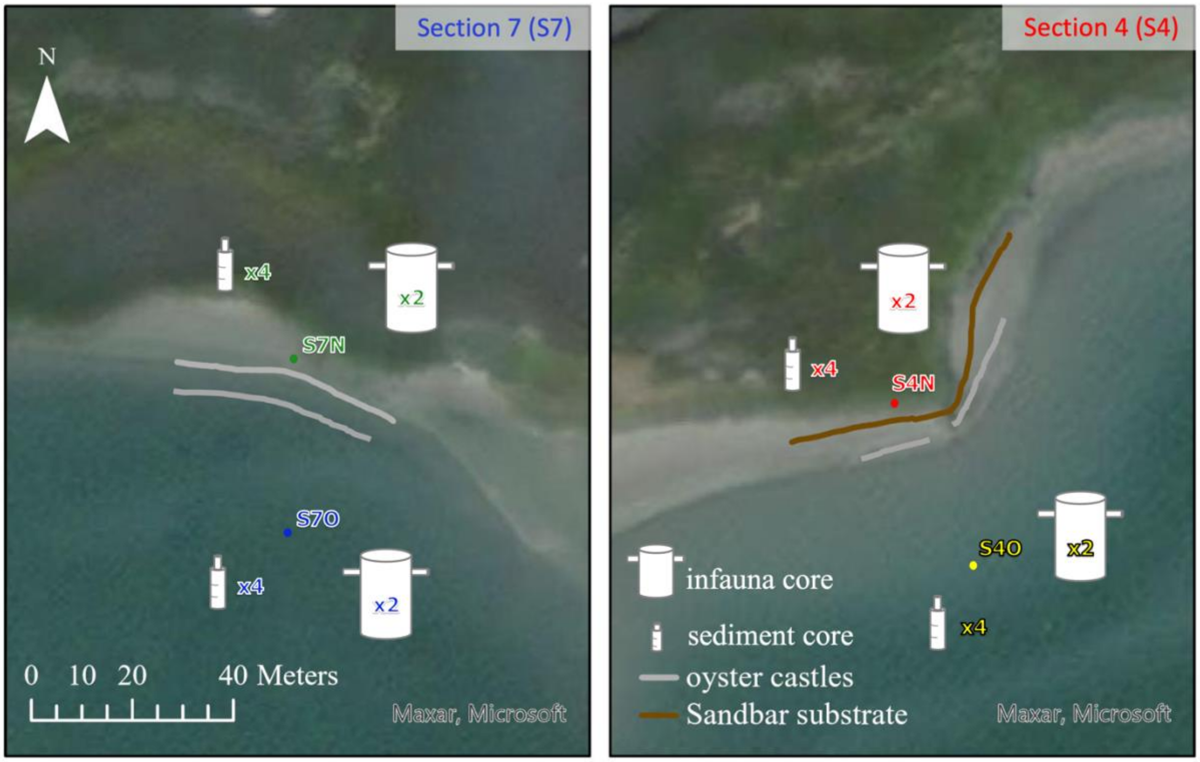

Figure 14. Locations and number of infauna and sediment samples collected from S4 and S7.

4.5 Infaunal properties

Information about infaunal diversity and abundance was gathered several times a year, coinciding with hydrodynamic measurements. Two random infaunal core samples were collected at each location S4N, S7N, S4O and S7O using a 15 cm wide in diameter by 15 cm deep PVC corer (Figure 14). Within 36 hours of collection, samples were passed through a 1 mm sieve and placed into jars containing 70% enthanol. Infauna were counted and sorted into broad taxonomic categories, including polychaetes (worms), small crustaceans (i.e., shrimp, amphipods), large crustaceans (i.e., crabs), gastropods (snails), bivalves and other (Hogan and Reidenbach, 2021). Samples were then placed in a drying oven at 60ºC for 48 hours to measure dry weight (g) and a muffle furnace at 500ºC for 6 hours to measure ash weight (g) (Rumohr, 1999; Hogan and Reidenbach, 2021). Organic matter content, or ash free dry weight (AFDW) of the infaunal samples was determined by the difference between sample dry and ash weight. Differences in infaunal abundance and AFDW between dates were determined using one-way ANOVAs and Tukey multiple comparisons tests. These protocol for monitoring sediment and infaunal characteristics associated with oyster reefs have already been put in place within the VCR (NFWF, 2019).

4.6 Marsh grass morphometrics

Cordgrass, *Spartina alterniflora*, samples were collected several times a year. Five random grass samples were clipped within a 25 cm x 25 cm quadrat at the marsh edge and at 10 m into the interior marsh at S4 and S7 (Figure 15). Following collection, grass densities and heights were recorded. Differences in marsh grass density and height between sections were determined using one-way ANOVAs.

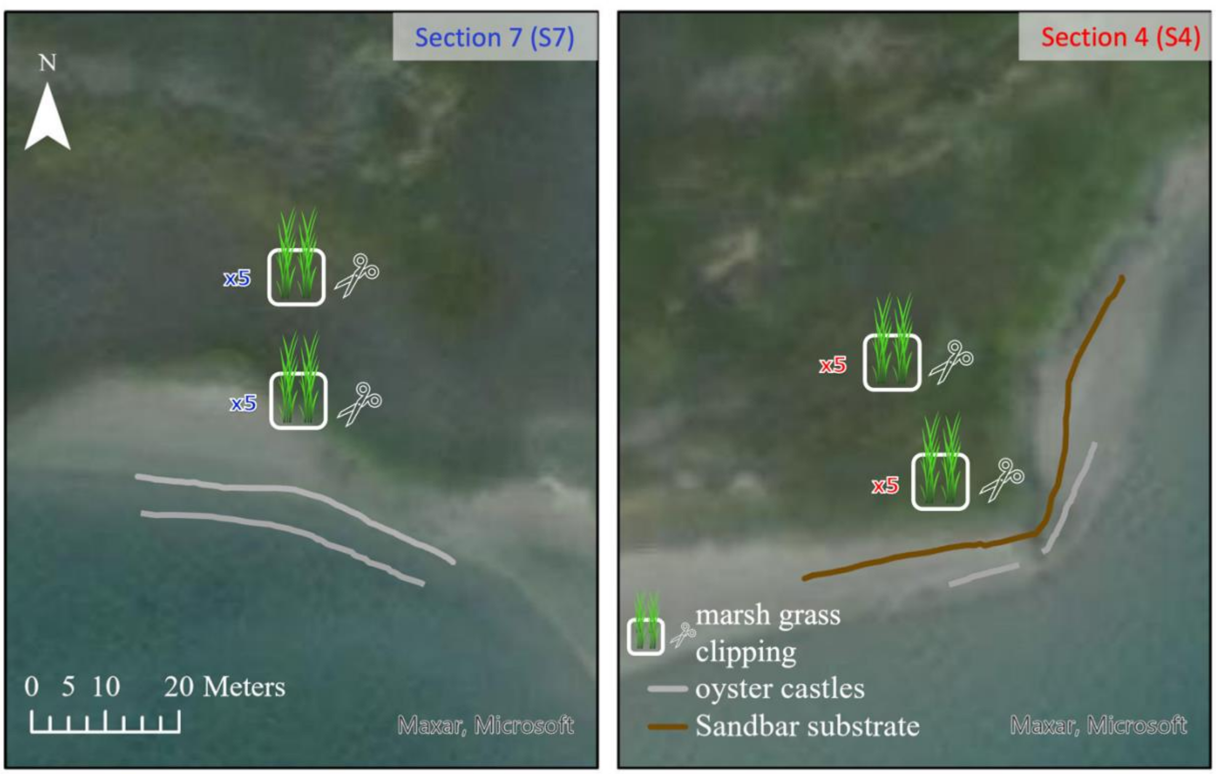

Figure 15. Locations of marsh grass clippings within quadrats.

4.7 Larval settlement

Larval settlement tiles (4 inch by 4 inch, or 0.0103 m^2, ceramic tiles) were deployed near the constructed oyster reefs at S4 and S7 during larval recruitment season. Two larval settlement tiles were positioned and attached vertically to each PVC pole using zip ties and inserted into the sediment in and around the constructed reefs (Figure 16 & 17). The tiles were situated 8 inches above the seafloor below the very tops of the reefs and within the interstices. At S7, three PVC poles were deployed between the oyster castles and one at a break in the oyster castle row containing bare sediment. At S4, three PVC poles were deployed just behind the row of oyster castles, three within the Sandbar substrate and one at a break in the oyster castle row containing bare sediment (Figure 17). After the five months, settlement tiles were collected, photographed and recruits (shell heights < 25 mm) were counted and their shell heights measured (Figure 16).

Figure 16. Two larval settlement tiles attached to a PVC pole using zip ties just after deployment (top left) and several months after deployment (top right). The front (bottom left) and back (bottom right) of one larval settlement tile after collection.

Figure 17. Larval settlement tile deployment locations.

4.8 Oyster densities and shell heights

Following the summer recruitment season, in-situ densities and shell heights of adult (shell heights > 25 mm) oysters were quantified on the constructed reefs at S4 and S7 using three 25 cm by 25 cm (0.0625 m^2) quadrats randomly placed on each reef (Figure 18). Images of quadrats were taken prior to counts (Figure 19). On the oyster castles, 10 random oysters were measured on the top tier and 5 on the bottom tier within each quadrat. On the Sandbar substrate, 15 random oysters were measured within each quadrat. Adult oysters were classified as those with shell heights greater than 25 mm. Average shell heights were measured for adult oysters. One-way ANOVAs were used to determined differences in adult and recruit oyster densities between substrates. A one-way ANOVA was also used to assess differences in adult shell heights between substrates, however the Kruskal-Wallis rank sum test, the non-parametric version of a one-way ANOVA, was used to assess differences in recruit shell heights between substrates. Quantification of recruitment and density is important to understand initial and long-term restoration success.

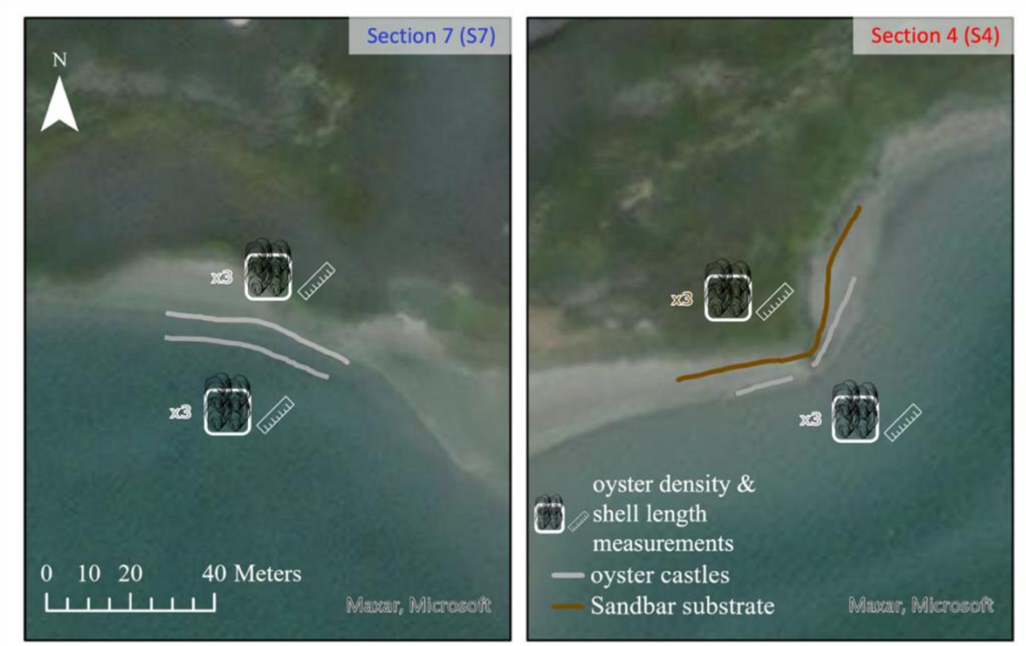

Figure 18. Locations of random oyster density and shell height measurements

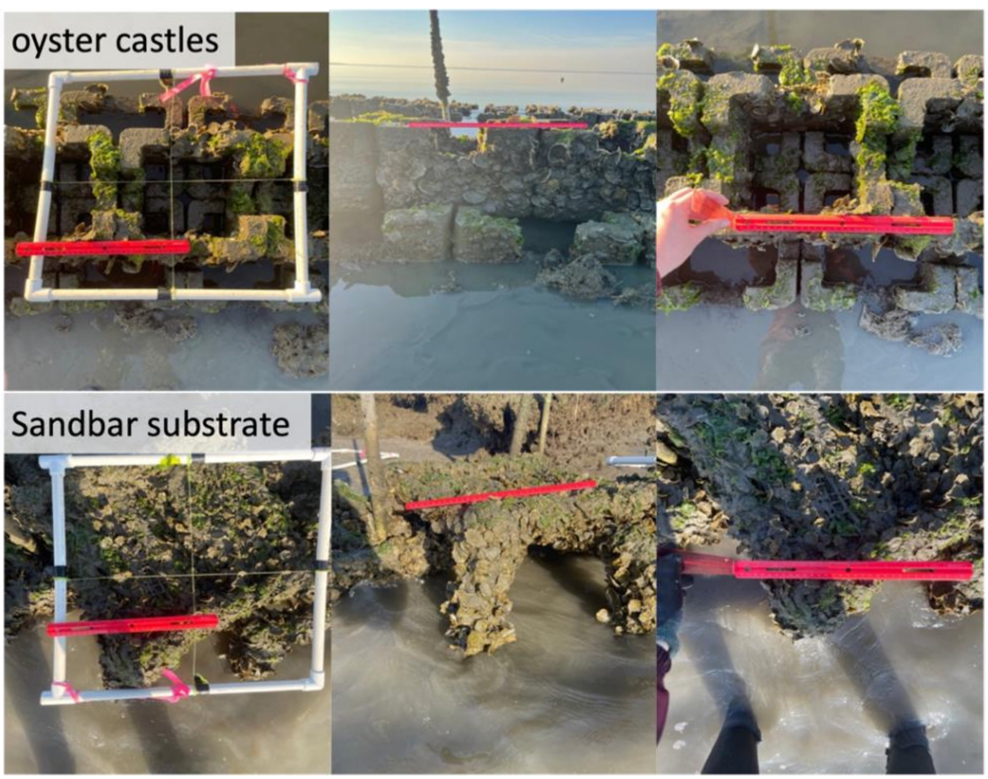

Figure 19. Images of quadrats used to collect oyster density and shell height measurements.

4.9 Statistical analysis

Shapiro-Wilk tests, Levene's tests, Q-Q plots and other visual examinations of residual spread were used to assess normality and heteroscedasticity. If necessary to meet ANOVA assumptions, factors were either log or square root transformed prior to testing. Non-parametric tests were used in place of ANOVAs if assumptions were not met.

5. Results

5.1 Site characteristics

Time series of water depth, significant wave height (*Hs*), turbidity, wind speed and direction for deployments are presented in Figures 20-23 and summarized in Tables 2 & 3. Winds at this location tended to most frequently originate from the south and northwest with occasional strong winds from the northeast (Figure 24). Current speeds averaged 0.045 m s^{-1} and the dominant flow axis was west-north-west/east-south-east in the small channel running parallel to the general orientation of the shoreline (Figure 25). On average, wave energy reaching S7 is significantly greater than that reaching S4 during deployment time periods in this study (Appendix 1).

Table 2. Mean, standard error (se) and max depth and *Hs* for each deployment location and period that were used in analysis.

Deployment	Location	Depth (mean +/- se/max)	*Hs* (mean +/- se/max)
Aug 31 – Sep 27, 2020	S4N	0.81 +/- 0.007/1.86	0.03 +/- 0.0006/0.19
	S4O	1.32 +/- 0.007/2.37	0.02 +/- 0.0005/0.20
	S7N	0.80 +/- 0.007/1.83	0.03 +/- 0.0005/0.19
	S7O	1.21 +/- 0.007/2.24	0.02 +/- 0.0005/0.21
Nov 27, 2020 – Jan 15, 2021	WC	1.27 +/- 0.006/2.32	0.01 +/- 0.0002/0.17
	BB	1.05 +/- 0.006/2.14	0.04 +/- 0.0008/0.51
Jun 25 – Jul 26, 2021	S4M	0.67 +/- 0.005/1.35	0.095 +/- 0.001/0.61
	S4O	1.38 0.006/2.13	0.072 +/- 0.001/0.53
	S7N	0.62 +/- 0.005/1.35	0.09 +/- 0.001/0.45
	S7O	1.37 +/- 0.005/2.09	0.08 +/- 0.001/0.54
Nov 11 – Dec 11, 2021	S4N	0.51 +/- 0.005/1.12	0.05 +/- 0.001/0.29
	S4M	0.77 +/- 0.005/1.33	0.05 +/- 0.001/0.31
	S4O	1.43 +/- 0.005/2.04	0.03 +/- 0.001/0.30
May 19 – Jun 13, 2022	S7N	0.66 +/- 0.006/1.36	0.06 +/- 0.001/0.33
	S7O	1.35 +/- 0.007/2.05	0.05 +/- 0.001/0.30

Table 3. Mean, standard error (se) and max wind speeds for each deployment period.

Start-End Date	Wind speed (mean +/- se/max)
Aug 31 – Sep 27, 2020	3.51 +/- 0.036/11.4
Nov 27, 2020 – Jan 15, 2021	2.43 +/- 0.023/14.8
Jun 25 – Jul 26, 2021	3.19 +/- 0.029/13.8
Nov 11 – Dec 11, 2021	2.31 +/- 0.024/8.51
May 19 – Jun 13, 2022	3.59 +/- 0.041/12.6

Mean water level (MWL), the average of depths recorded by offshore sensors, was 1.34 and 1.31 m for S4 and S7, respectively. Mean depths relative to MWL for sensors locations, reef crests and marsh edges are summarized in Figure 26 and Table 4. The offshore and nearshore reef crests at S7 averaged 0.81 and 0.95 m high (0.35 – 0.5 m below MWL). The offshore and nearshore reef crests at S4 averaged 0.88 and 1.18 m high (0.35 – 0.6 below MWL).

Table 4. Depths relative to mean water level (MWL), average of depths recorded at offshore sensors, for each section. MWL for S4 and S7 were 1.34 and 1.31 m, respectively.

Location	Mean depth relative to MWL (m)
S4N	-0.26
S4M	-0.56
S4O	-1.34
S7N	-0.59
S7O	-1.31
S4 marsh edge	0.54
S7 marsh edge	0.25
S4 nearshore crest	-0.16
S4 offshore crest	-0.46
S7 nearshore crest	-0.36
S7 offshore crest	-0.5

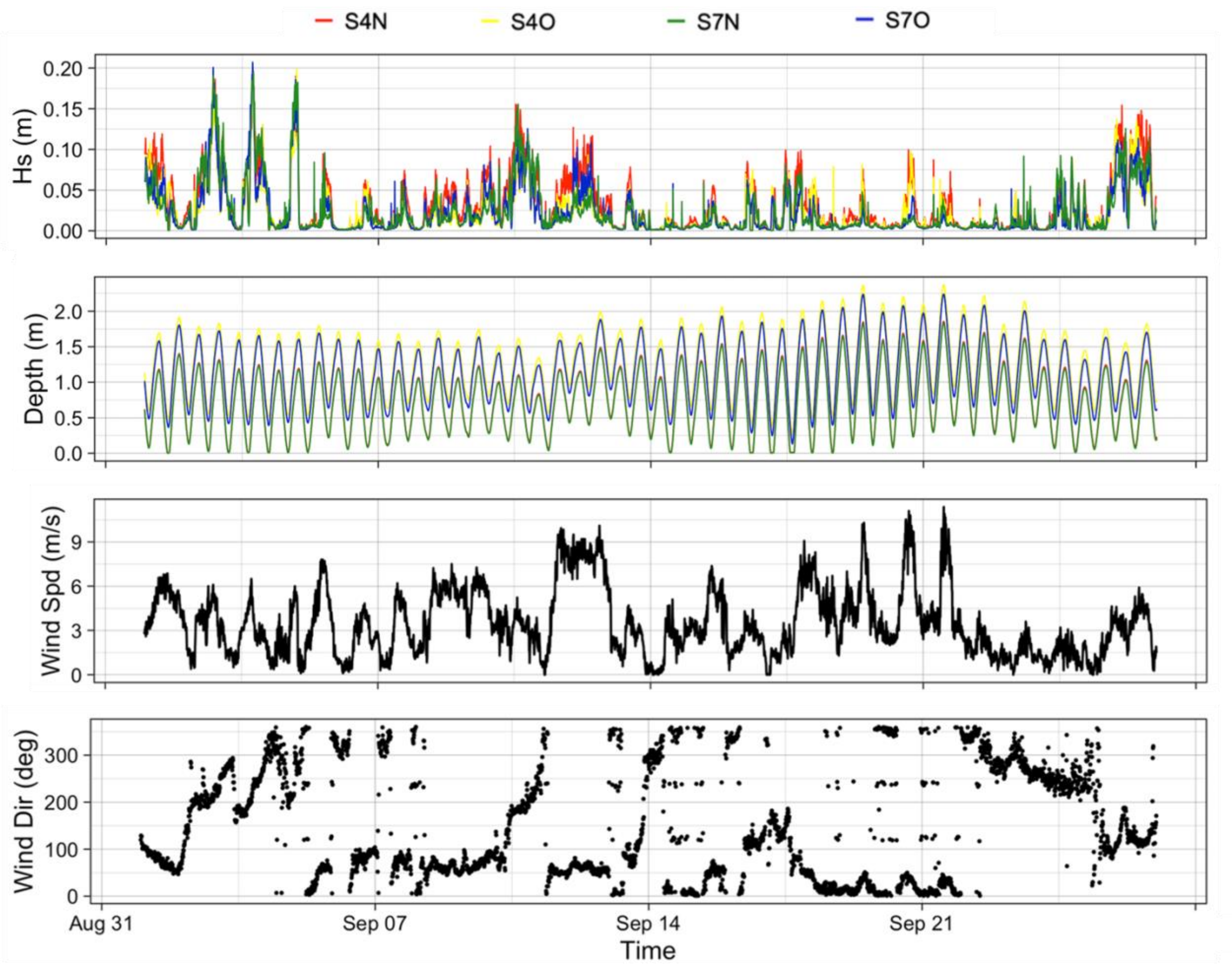

Figure 20. *Hs*, water depth, wind speed and direction from a pre-construction deployment August 31 – September 27, 2020 before data filtering.

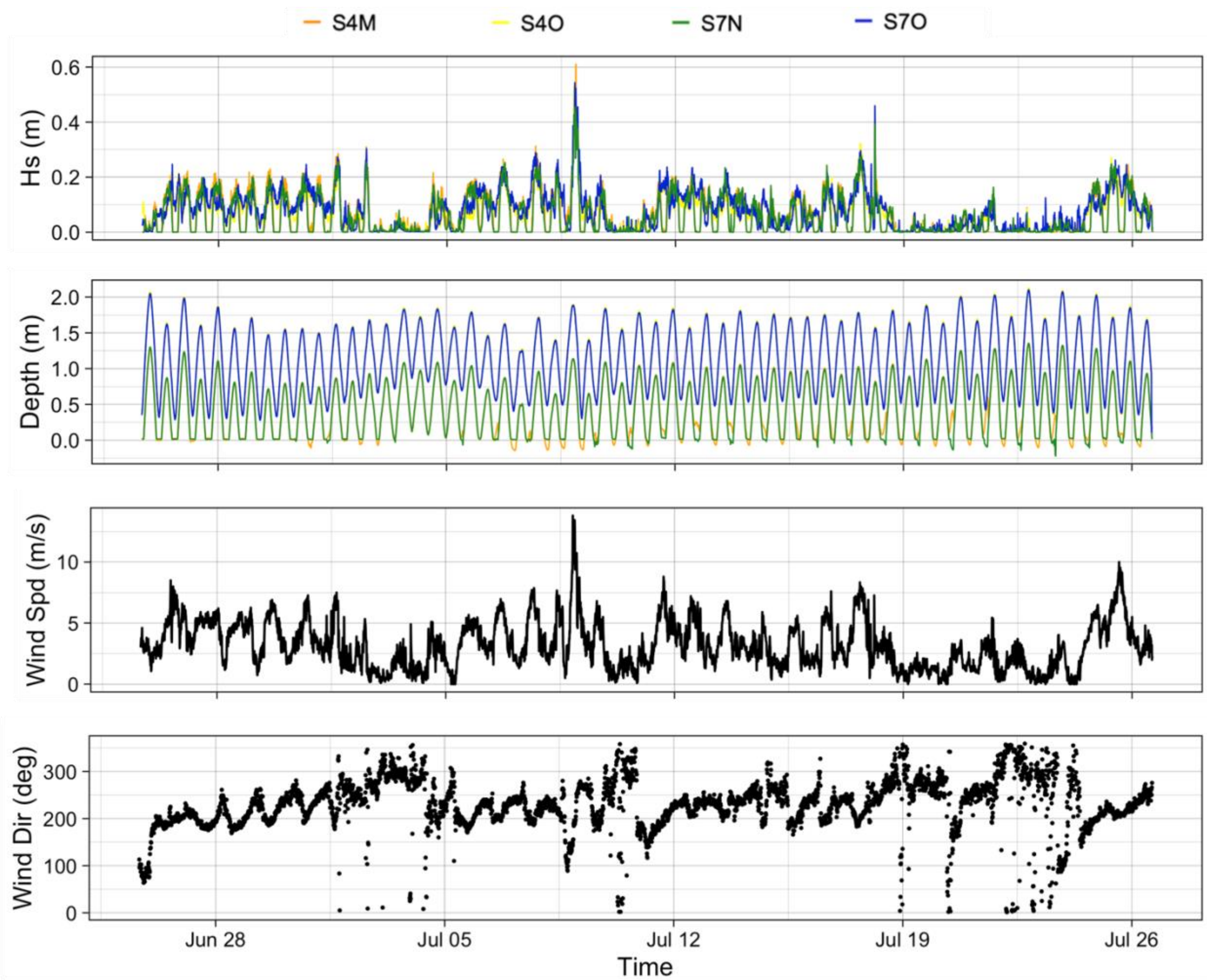

Figure 21. *Hs*, water depth, wind speed and direction from a post-construction deployment June 25 – July 26, 2021 before data filtering.

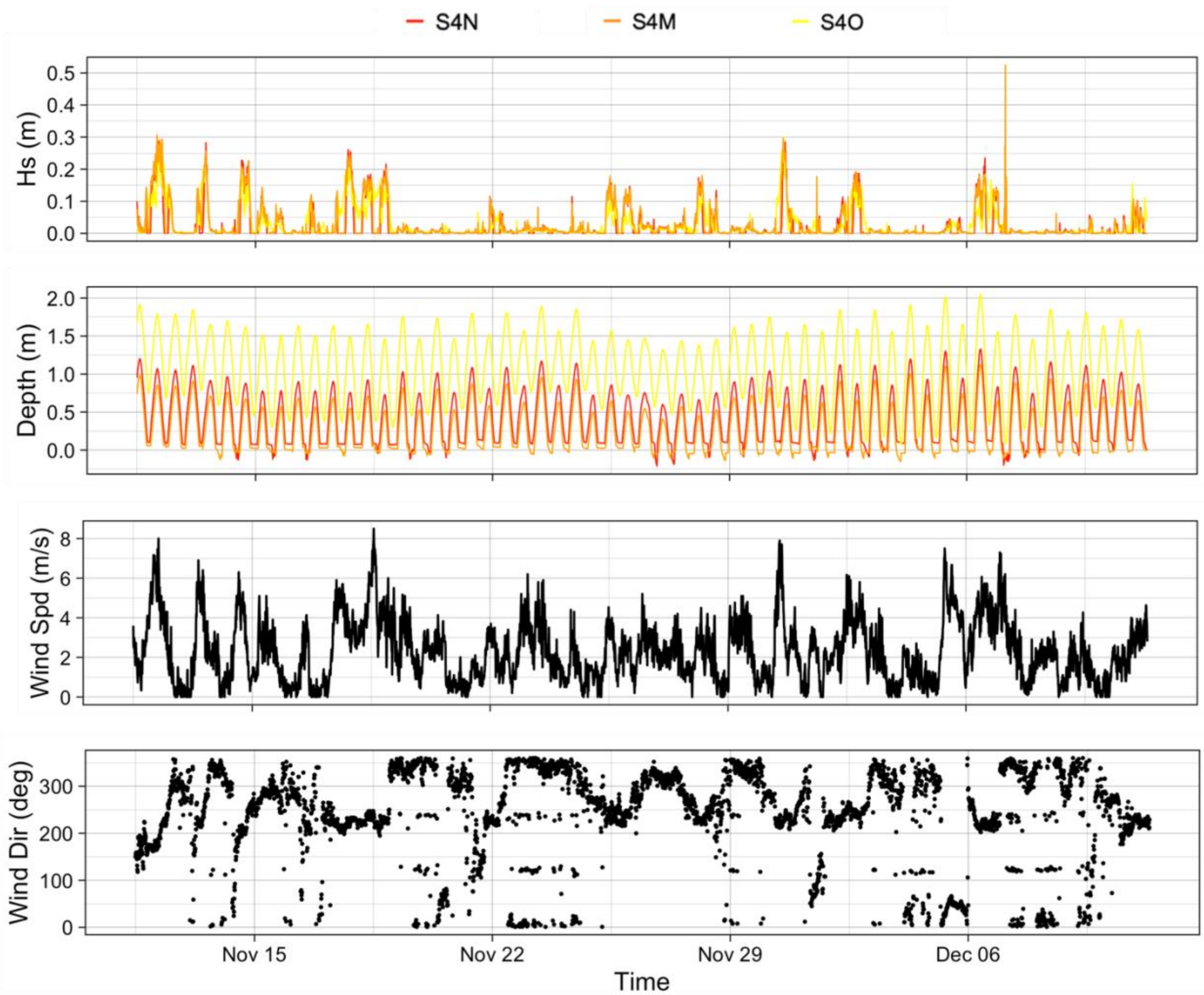

Figure 22. *Hs*, water depth, wind speed and direction from a post-construction deployment November 11 – December 11, 2021 before data filtering.

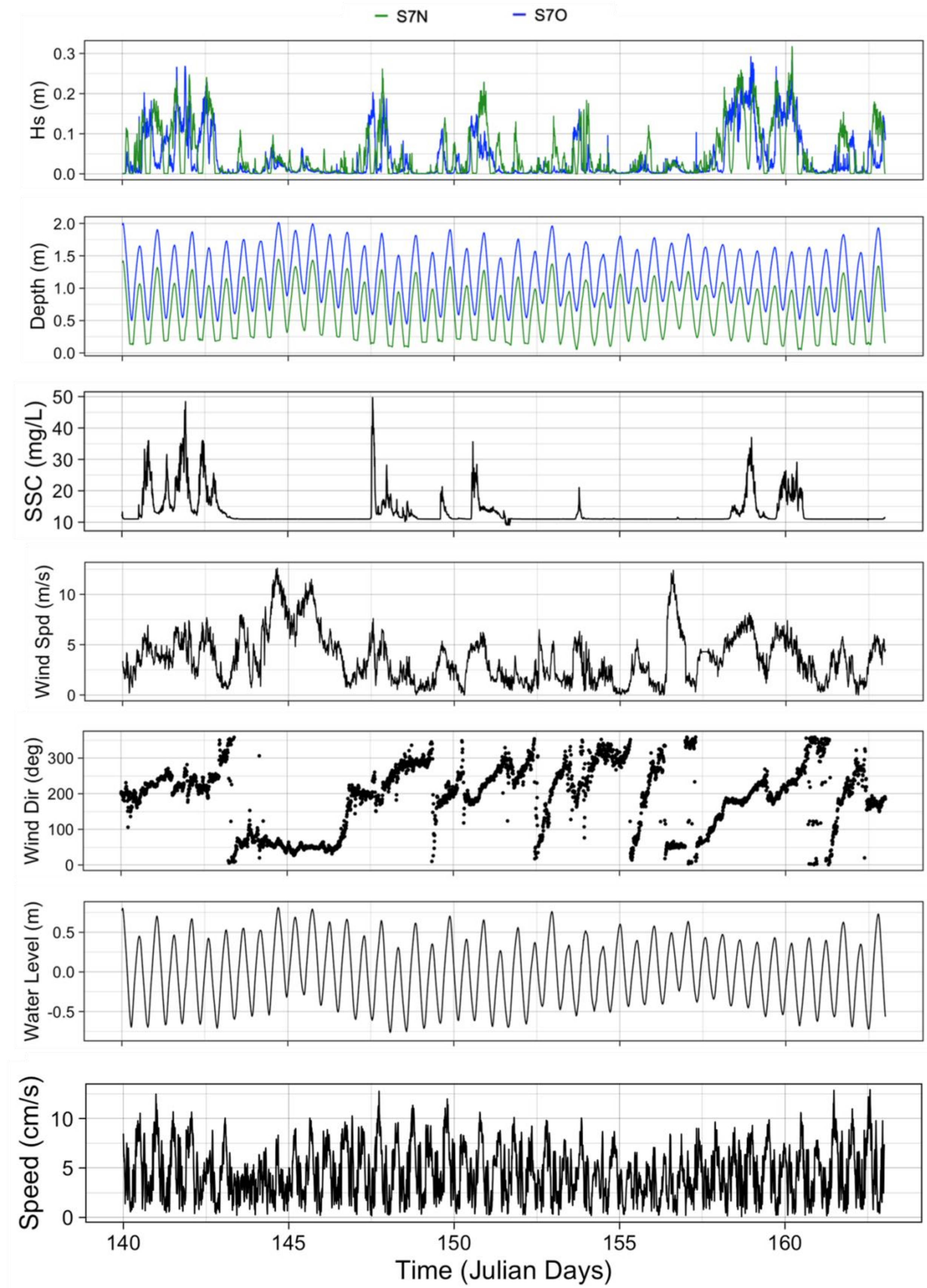

Figure 23. Water depth, *Hs*, turbidity, wind speed and direction, water level and speed from a post-construction deployment May 19 – June 13, 2022.

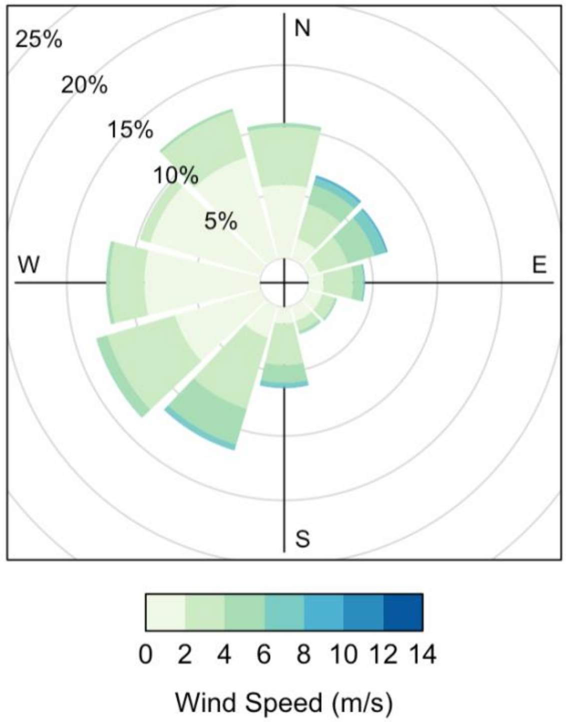

Figure 24. Wind rose of data recorded at the NOAA station (Station ID: 8631044) in Wachapreague, VA from September 1, 2020 – April 1, 2022.

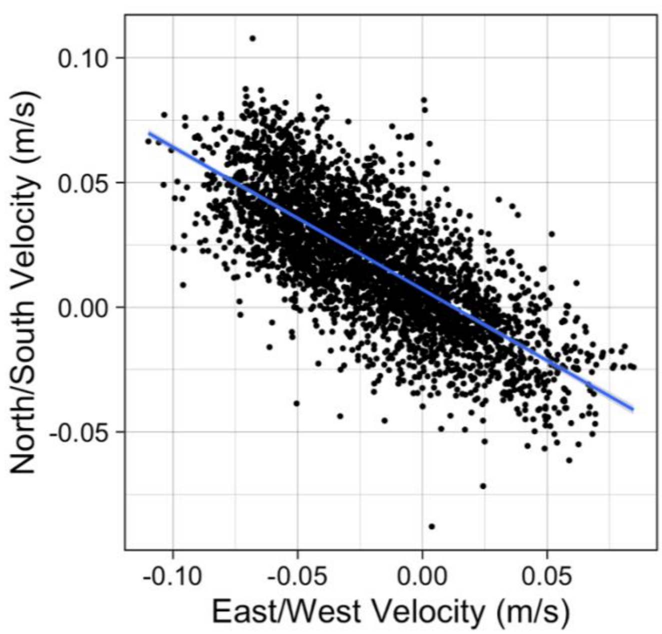

Figure 25. North/South and East/West velocities collected at 0.5 m above the seafloor using an Aquadopp from May 19 – June 13, 2022. Data reveal a dominant west-north-west/east-south-east flow direction.

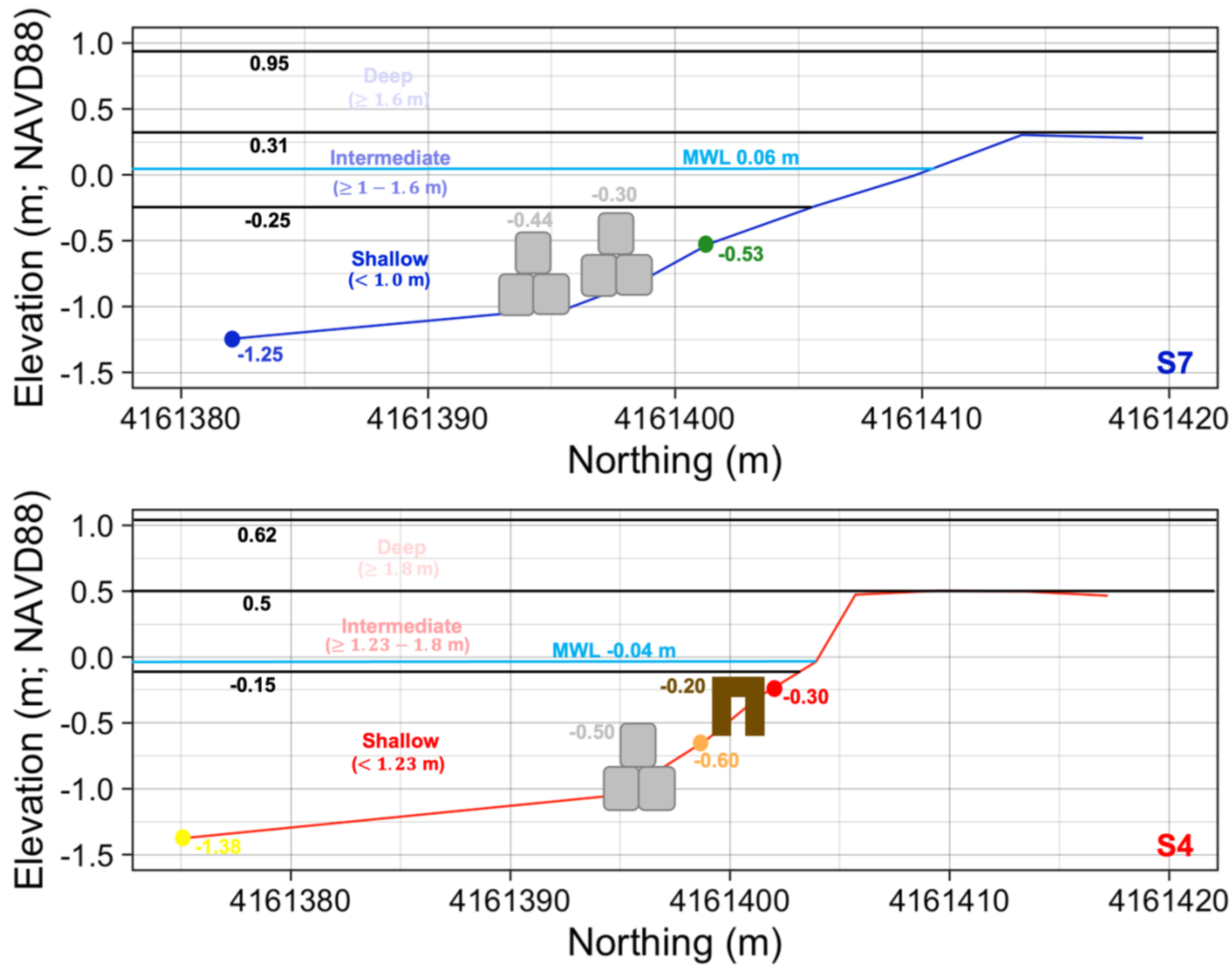

Figure 26. Cross sectional profiles of Sections 4 and 7 including elevations, relative to NAVD88, of sensor locations, reef crests and water depths split into shallow, intermediate and deep. Mean water level (MWL), the average depth recorded at each offshore sensor, is also indicated (Table 4).

5.2 Hydrodynamics

Mean significant wave heights during all deployments ranged from 0.022 to 0.085 m, with the maximum wave height reaching 0.61 m at S4M in July 2021. Wave heights and turbidity increased due to elevated episodic wind speeds, especially when winds blew from the southwest (Figures 20-23, Tables 2 & 3). Pre-construction (Aug 31-Sep 27, 2020), wave heights were significantly higher at nearshore than offshore sites at both S4 and S7, suggesting wave amplification as water depths decrease close to the marsh edge (Figures 27 & 28, Table 2, Appendix 2).

Post-construction, wave heights at S7 were significantly lower at nearshore sites than offshore sites in shallow water ($\leq 0.05\ m$ above nearshore reef crest), suggesting some degree of wave attenuation. However, in intermediate and deep water ($> 0.5\ m$ above nearshore reef crest) nearshore wave heights tended to be greater than offshore wave heights, indicating no change from pre-construction wave activity (Figures 27 & 28, Table 2, Appendix 2). At S4, nearshore and middle wave heights were significantly higher than offshore wave heights at all water depths, even post-construction (Figures 27 & 28, Table 2, Appendix 2).

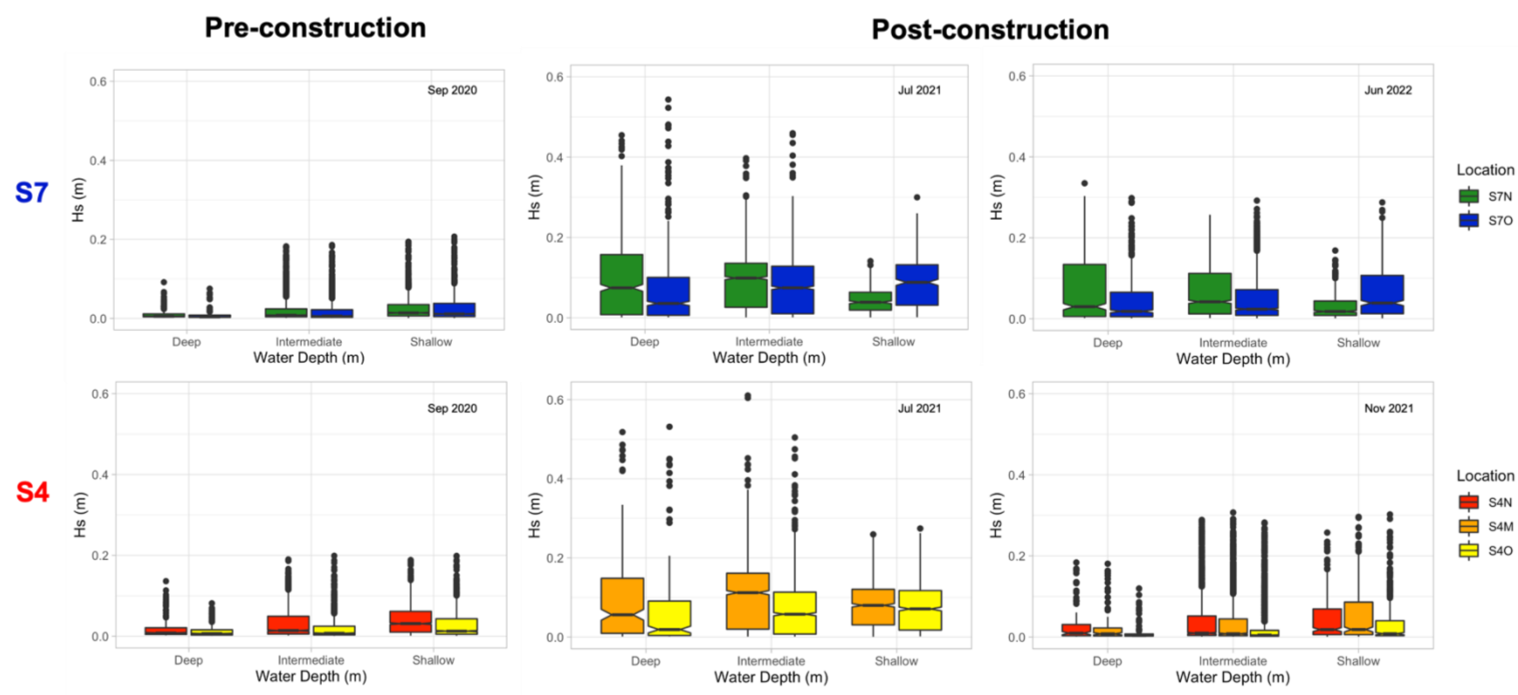

Figure 27. Box-plots of significant wave heights (*Hs*) at S4 (red) and S7 (blue) during multiple deployments. *Hs* are separated into shallow ($\leq 0.05\ m$ above nearshore reef crest), intermediate ($> 0.5\ m$ above nearshore reef crest to water depth at marsh platform) and deep ($\geq$ water depth of marsh platform) water conditions and colored by location. Statistics are summarized in Appendix 2.

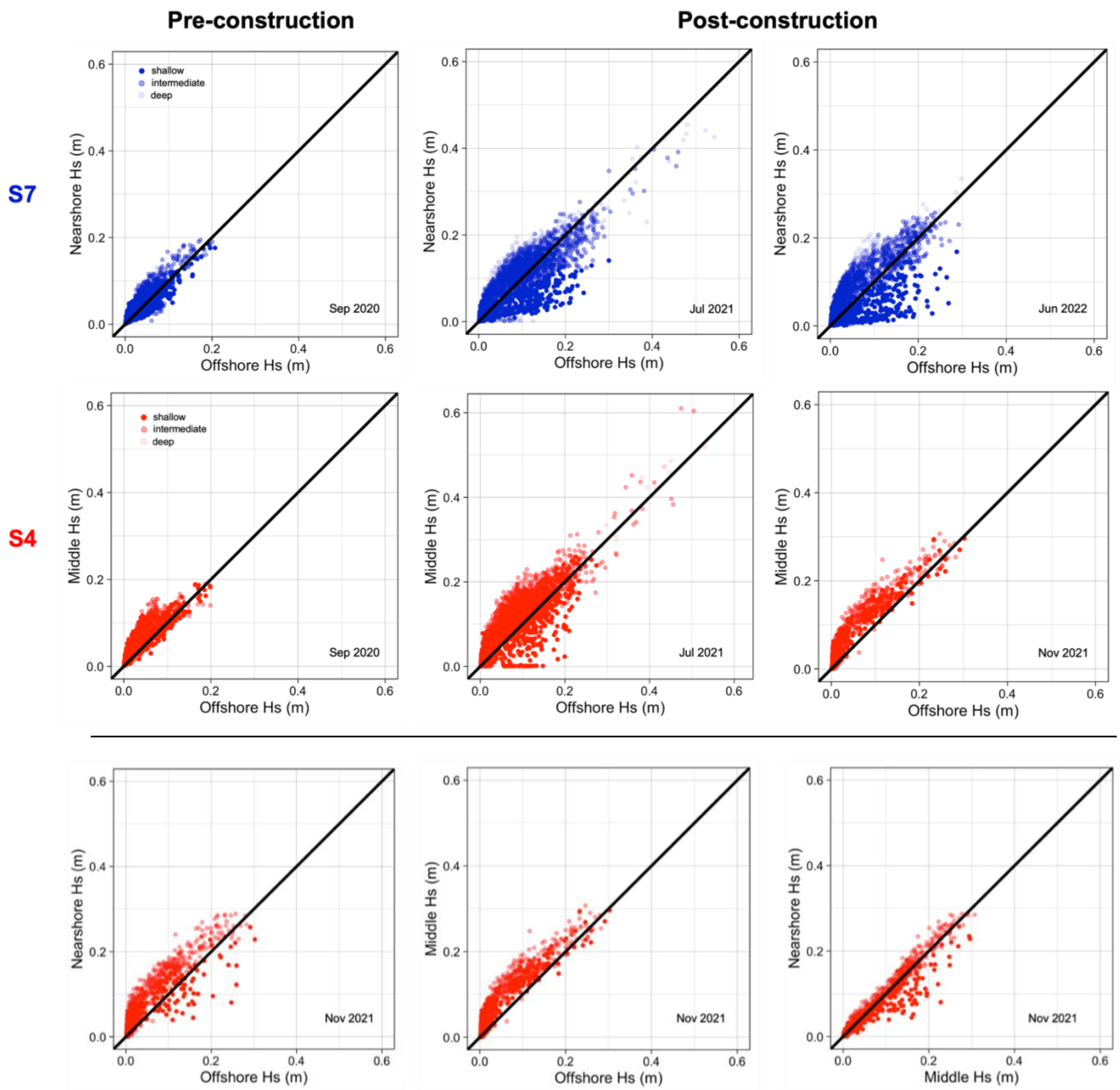

Figure 28. Scatter plots of significant wave heights (*Hs*) at offshore, middle and nearshore locations at S4 (red) and S7 (blue) during multiple deployments (above) and during the November 11 – December 11, 2021 deployment (below). *Hs* are separated into shallow ($\leq 0.05\ m$ above nearshore reef crest), intermediate ($> 0.5\ m$ above nearshore reef crest to water depth at marsh platform) and deep ($\geq$ water depth of marsh platform) water conditions.

5.3 Sediment accumulation

The mass of sediment accumulated on sediment tiles placed on the marsh surface did not differ significantly between Sections 4 and 7 nor between locations on the marsh (edge, middle, interior) (Figure 29, Table 5, Appendix 3-4). However, photographs of Sandbar substrate at Section 4 indicate some degree of sediment accumulation in and around the substrate from August to September, 2021 (Figure 30).

Table 5. Sample size, mean, standard deviation and error of sediment accumulation on marsh platform by date and location on marsh.

		Sediment accumulation (g per cm2)		
Section	n	mean	sd	se
4	17	0.01	0.009	0.002
7	15	0.006	0.007	0.002

		sediment accumulation (g per cm2)		
Location	n	mean	sd	se
edge	8	0.005	0.01	0
middle	12	0.01	0.01	0
interior	12	0.01	0.01	0

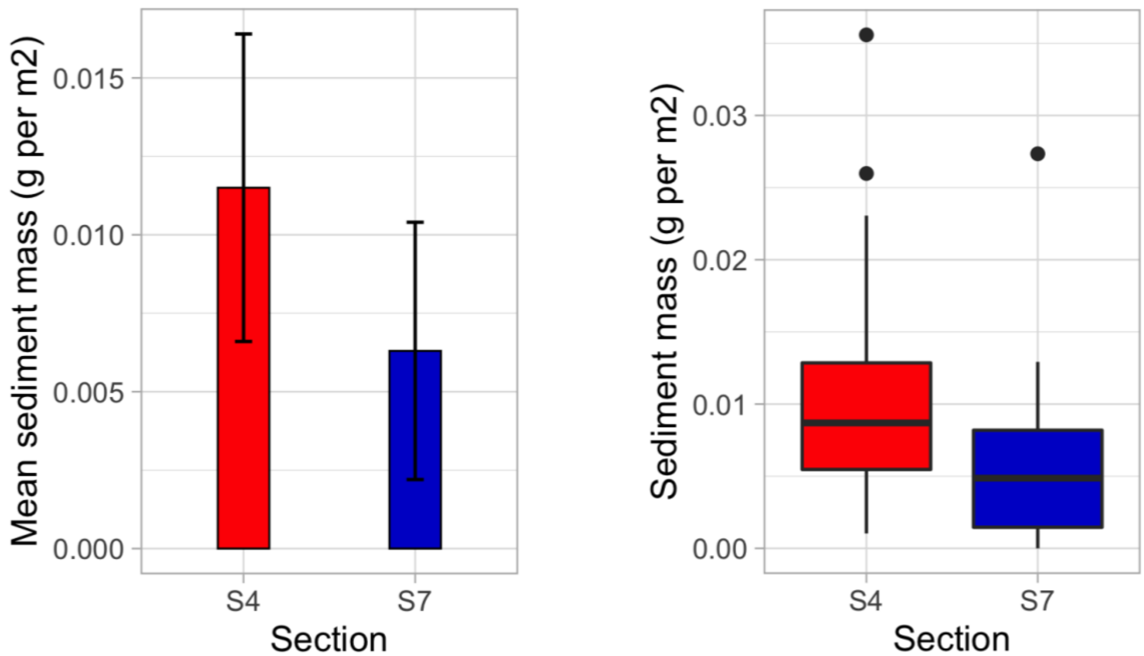

Figure 29. Average accumulated sediment mass with 95% confidence intervals (left) and spread of accumulated sediment masses (right) on the marsh platforms at Sections 4 and 7 grouped from measurements taken in Jul and Sep 2021 and Jul 2022 (Table 5). No significant differences were found between sections. Results from a 1-way ANOVA are summarized in Appendix 3).

Figure 30. Sediment accumulating in and around the Sandbar substrate at Section 4 from August – September, 2021.

5.4 Marsh edge morphology

In-situ measurements of 22 erosion pins on the Bradford Bay (exposed) side of the marsh island (Figure 12) in June 2022, a year after deployment, showed that constructed oyster reefs significantly reduced rates of marsh edge erosion. There was a range of 0 to 140 cm (mean +/- SD cm; 53 +/- 48) of marsh retreat at locations behind oyster reefs and a range of 74 to 229 cm (124 +/- 52) of retreat at locations lacking restored reefs (Figure 31, Table 6, Appendix 5). Locations behind oyster reefs where large amounts of retreat occurred were generally areas of the marsh island that are more exposed, such as the eastern and western tips, and have seen heavy erosion rates in the past (Figures 2 & 31). There was no significant difference in marsh retreat between erosion pin measurements at S4 and S7 (Appendix 6).

Table 6. Sample size, mean, standard deviation and error of distance eroded by presence of reefs.

		Distance eroded (cm)		
	n	mean	sd	se
no reef	9	124	52	17
reef	13	53	48	13

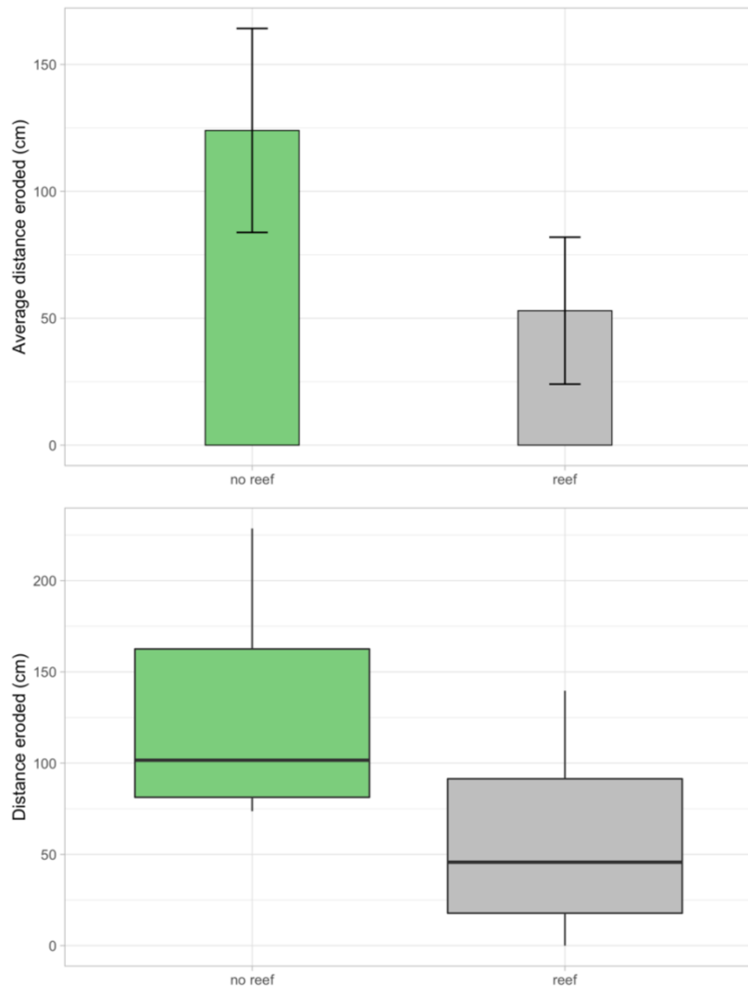

Figure 31. Average distance eroded (top) and spread of distances eroded (bottom) at locations with and without oyster reefs over the course of one year. Error bars represent 95% confidence intervals surrounding the means. Significant differences were found between locations with and without oyster reefs. Results from a 1-way ANOVA are summarized in Table 6 and Appendix 5.

5.5 Marsh grass morphometrics

A summary of vegetation characteristics for Sections 4 and 7 is presented in Table _. Marsh grass stem height and density were significantly different between sections. Section 4 had the greatest mean stem densities while Section 7 had the greatest mean stem heights (Figure 32, Table 7, Appendix 7-8).

Table 7. Sample size, mean, standard deviation and error of stem height and density by section.

	Section 4		Section 7	
	Height (cm)	Density (per cm2)	Height (cm)	Density (per cm2)
n	10	10	10	10
mean	17.31	0.13	21.93	0.09
sd	3.61	0.05	4.23	0.03
se	1.14	0.01	1.34	0.01

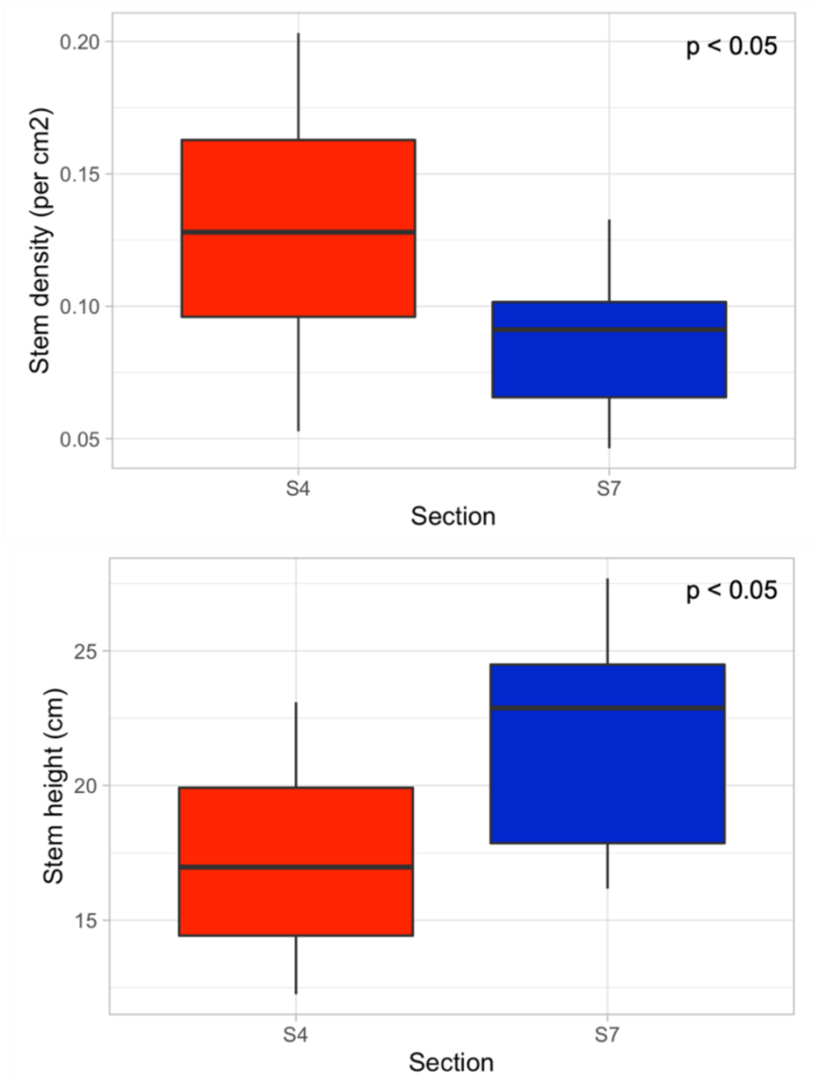

Figure 32. Marsh grass stem height (left) and density (right) measurements for Sections 4 and 7 from July 2021. Significant differences between sites are indicated by $p < 0.05$. Results from 1-way ANOVAs are summarized in Table 7 and Appendix 7-8.

5.6 Sediment and infaunal properties

One-way ANOVAs revealed no significant difference in percent sediment organic matter content or infaunal AFDW between sampling dates or site location ($p > 0.05$) (Figures 33 & 34, Tables 8 & 9, Appendix 9-12). There was also no significant difference in infaunal count between sampling dates or site location ($p > 0.05$) (Figure 35, Table 9, Appendix 13-14). The differences in total infaunal abundance and AFDW, though insignificant, were driven by worm abundance (Figures 33 & 34, Table 9).

Table 8. Sample size, mean, standard deviation and error of % sediment organic matter content by date.

% sediment organic matter content				
Date	n	mean	sd	se
Sep 2020	8	5.93	2.75	0.97
Jul 2021	8	2.73	2.19	0.77
Oct 2021	8	3.6	2.98	1.07
Jul 2022	8	2.45	2.65	0.94

Table 9. Mean infaunal abundance, biomass (g of ash free dry weight) and percent organic matter +/- standard error for 8 samples collected each date.

Date	Abundance (count)	AFDW (g)	OM (%)
Sep 2020	3.6 +/- 1.35	0.05 +/- 0.02	65% +/- 12
Jul 2021	2.14 +/- 0.85	0.05 +/- 0.03	55% +/- 15.4
Oct 2021	1.6 +/- 0.63	0.17 +/- 0.14	25% +/- 12.2

Sediment grain size samples revealed no significant differences in median grain size, or d_{50}, between sites (S4 and S7) nor between dates (July and October 2021). However, nearshore d_{50}s were significantly coarser than offshore sediments at both sites (Figure 36, Table 10, Appendix 15-17).

Table 10. Grain size attributes of sediment samples for each sampling date.

July 2021				
Average	S4N	S4O	S7N	S7O
Median grain size (um)	499	28	310	45
Percent sand	99.75	18	74	27.3
Percent silt	0.13	68	19.7	67
Percent clay	0.12	14	63	5.7

October 2021				
Average	S4N	S4O	S7N	S7O
Median grain size (um)	518	46	412	38
Percent sand	99.99	37	85.5	22
Percent silt	0.01	53	11	69
Percent clay	0	10	3.5	9

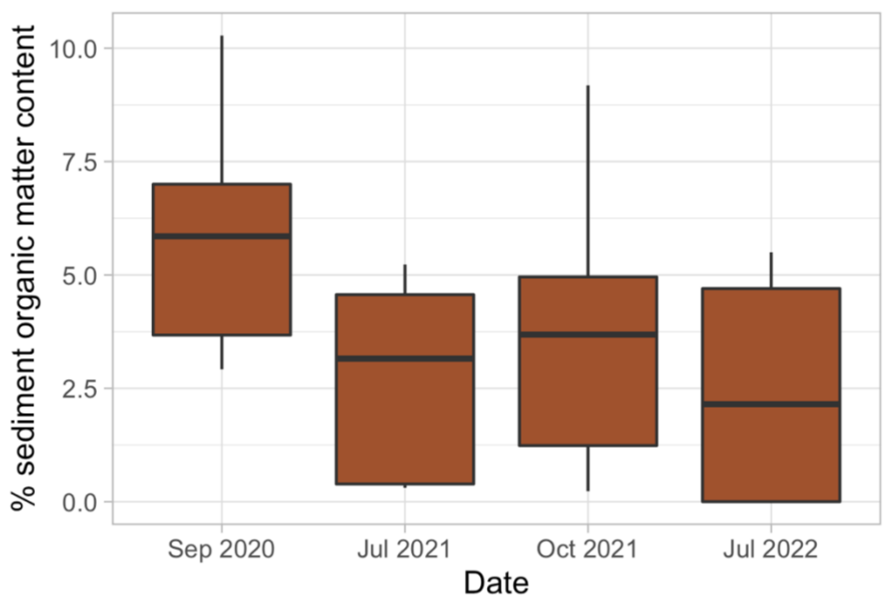

Figure 33. Percent sediment organic matter content pre- (Sep 2020) and post- (Jul and Oct 2021, Jul 2022) reef construction. No significant differences were found between dates. Results from a 1-way ANOVA are summarized in Table 8 and Appendix 9.

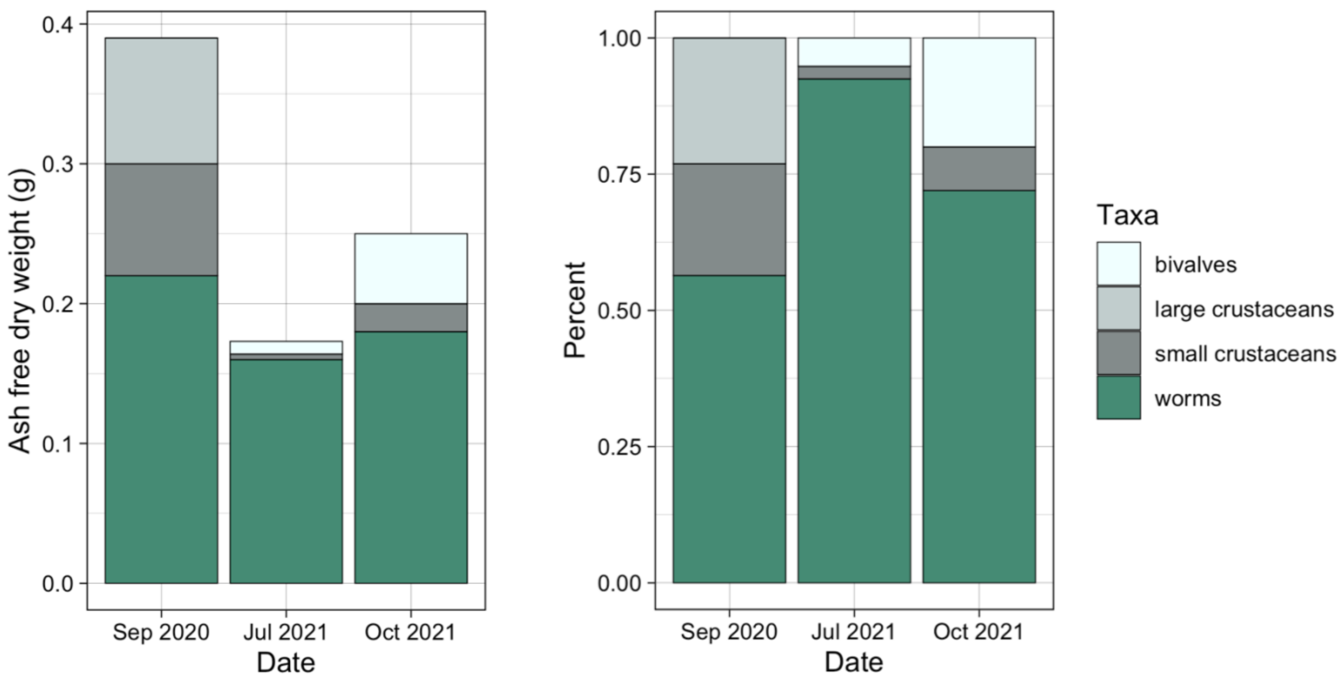

Figure 34. Infaunal ash free dry weight (AFDW) and percentage of AFDW for each taxon from the combined 8 samples for each date. No significant differences were found in AFDW between dates. Results from a 1-way ANOVA are summarized in Table 9 and Appendix 11.

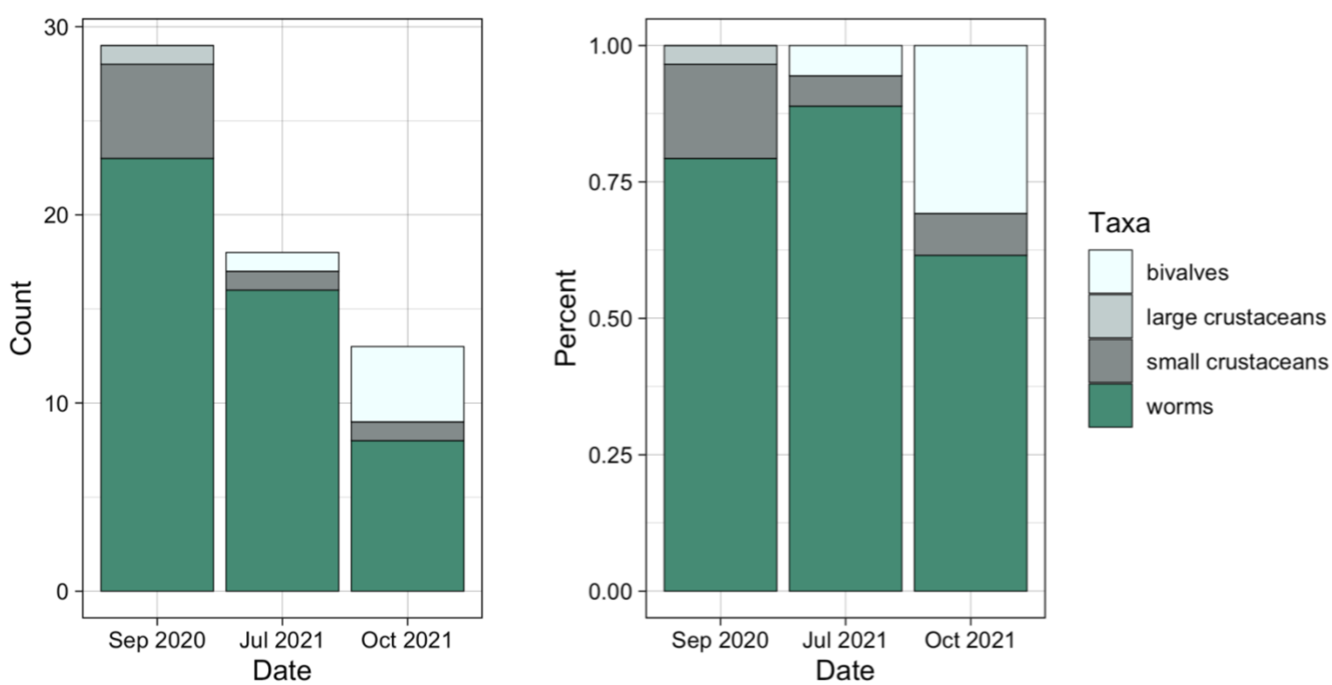

Figure 35. Infaunal abundance (count) and percentage of each taxon from the combined 8 samples for each date. No significant differences were found in count between dates. Results from a 1-way ANOVA are summarized in Table 9 and Appendix 13.

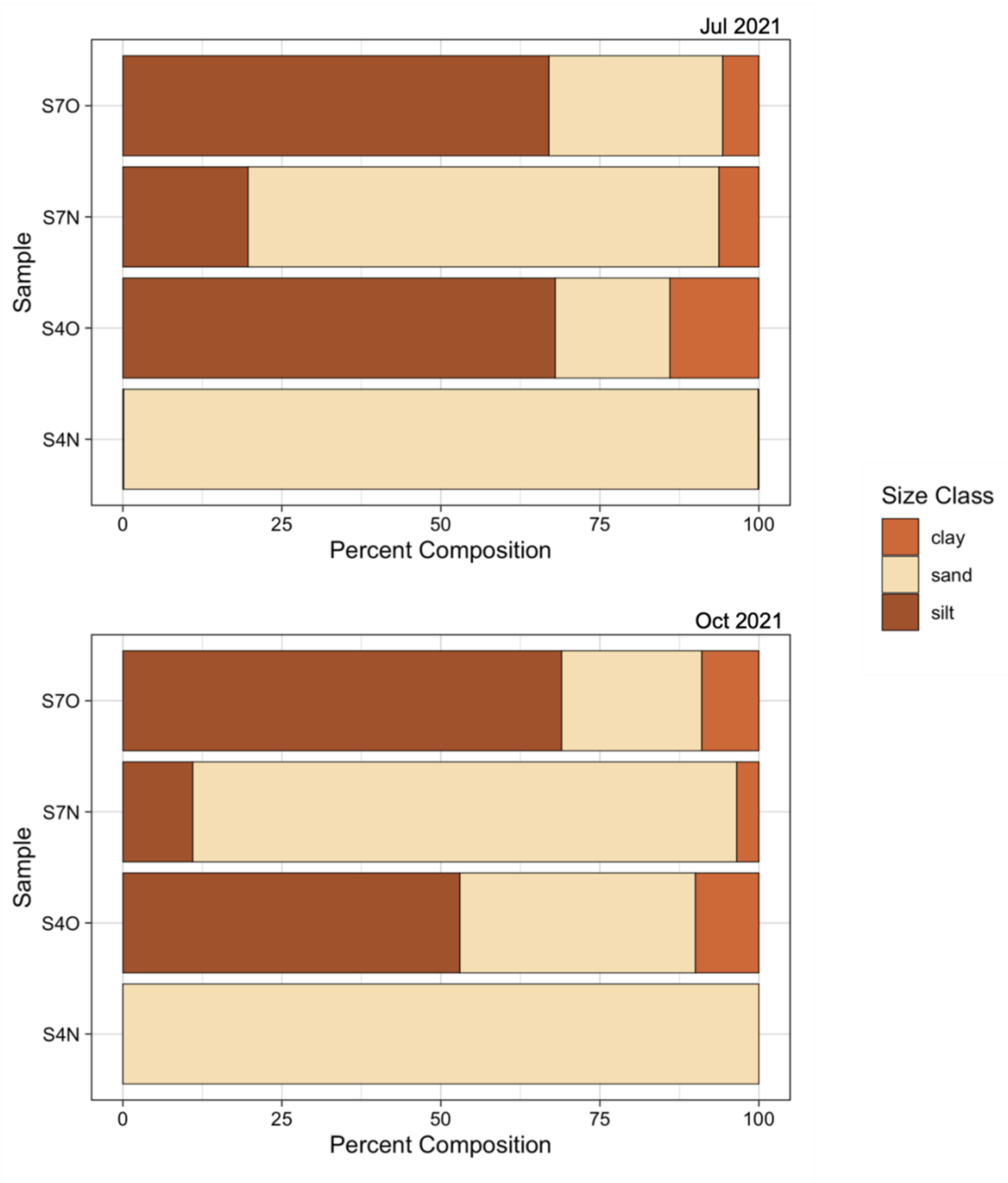

Figure 36. Grain size distributions for study samples from July (upper) and October 2021 (lower). Sand, silt and clay represent the following grain size ranges, respectively: > 63 um, 4-63 um and < 4 um. No significant differences were found in median grain sizes between sites or dates. Median grain sizes and percentages of size classes are summarized in Table 10 and results from a 1-way ANOVA are summarized in Appendix 16-17.

5.7 Oyster densities and shell heights

Reef crest elevations from August 2021 for each substrate and section are summarized in Table 11. On average, Sandbar substrate is located at a higher elevation (µ +/- SD m (NAVD88); -0.188 +/- 0.031) than oyster castles (-0.392 +/- 0.028), but both are still within the suitable elevation range for larval recruitment previously measured in the VCR (-0.81 to 0 m, NAVD88) (Tedford and Castorani, 2022; Hogan and Reidenbach, 2019).

Table 11. Mean and standard deviation of reef crest elevations by section and substrate.

Section	Substrate	Reef crest elevations mean +/- sd (m; NAVD88)
4	Sandbar	-0.188 +/- 0.031
4	oyster castles	-0.449 +/- 0.014
7	oyster castles – nearshore row	-0.286 +/- 0.045
7	oyster castles – offshore row	-0.441 +/- 0.026

The artificial reef designs in this study successfully fostered larval recruitment and oyster growth (Figure 37). During the first recruitment season in 2021, adult oyster densities (µ +/- SD per m^2; 1,324 +/- 739) were lower and less variable than recruit densities (4,664 +/- 1468) (Figure 38, Table 12). In 2022, adults (2,139 +/- 677) exceeded recruit (855 +/- 703) densities (Table 12). There was no significant difference in recruit densities between substrates for either sampling year. However, there was a significant difference in adult densities between substrates in 2021 and 2022 (Figure 39, Table 13, Appendix 18-19). In 2021, adult oyster densities for oyster castles and Sandbar substrate were 988 +/- 382 (µ +/- SD per m^2) and 2331 +/- 629, respectively (Table 13). By 2022, adult oyster densities on oyster castles and Sandbar substrate increased to 1812 +/- 381 (µ +/- SD per m^2) and 3120 +/-111, respectively (Table 13). Overall densities for each life stage are summarized in Table 12.

Table 12. Sample size, mean, standard deviation and error of all oyster densities by life stage per m^2 for each sampling year.

2021

Life stage	n	mean	sd	se
Adults	12	1324	739	213
Recruits	12	4664	1468	424

2022

Life stage	n	mean	sd	se
Adults	12	2139	677	195
Recruits	16	855	703	176

Table 13. Sample size, mean, standard deviation and error of oyster densities by substrate and life stage per m^2 for each sampling year.

2021

	Oyster Castles		Sandbar	
Life stage	Adults	Recruits	Adults	Recruits
n	9	10	3	2
mean	988	4414	2331	5917
sd	382	1485	629	274
se	127	470	363	194

2022

	Oyster Castles		Sandbar	
Life stage	Adults	Recruits	Adults	Recruits
n	9	12	3	4
mean	1812	776	3120	1091
sd	381	595	111	1037
se	127	172	64	158

Results from a one-way ANOVA revealed a significant difference ($p < 0.001$) in adult shell heights between substrates in 2021, but not in 2022 (Appendix 20). In 2021, adult shell heights for oyster castles and Sandbar substrate were 33.4 +/- 6.94 (µ +/- SD mm) and 31.4 +/- 5.23, respectively (Table 14). By the second sampling year (2022), adult shell heights for oyster castles and Sandbar substrate increased to 46.7 +/- 11.9 (μ +/- SD mm) and 48.5 +/- 9.73, respectively (Table 14).

There was also a significant difference ($p < 0.001$) in recruit shell heights between substrates for both sampling years (Appendix 21). In 2021, recruit shell heights for oyster castles and Sandbar substrate were 9.51 +/- 4.8 (μ +/- SD mm) and 12.28 +/- 6.21, respectively (Table 14). In 2022, recruit shell heights for oyster castles and Sandbar substrate decreased to 0.8 +/- 0.38 (μ +/- SD mm) and 0.52 +/- 0.19, respectively (Table 14). Overall shell heights for each life stage are summarized in Table 15.

Table 14. Sample size, mean, standard deviation and error of shell heights (mm) by substrate and life stage for each sampling year.

2021

	Oyster Castles		Sandbar	
Life stage	Adults	Recruits	Adults	Recruits
n	89	444	20	121
mean	33.4	9.51	31.4	12.28
sd	6.94	4.8	5.23	6.21
se	0.74	0.23	1.17	0.56

2022

	Oyster Castles		Sandbar	
Life stage	Adults	Recruits	Adults	Recruits
n	135	109	45	41
mean	46.7	0.8	48.5	0.52
sd	11.9	0.38	9.73	0.19
se	1.02	0.04	1.45	0.03

Table 15. Sample size, mean, standard deviation and error of all shell heights (mm) by life stage for each sampling year.

2021

Life stage	n	mean	sd	se
Adults	109	33.03	6.69	0.64
Recruits	565	10.11	5.26	0.22

2022

Life stage	n	mean	sd	se
Adults	180	47.2	11.4	0.85
Recruits	150	0.72	0.36	0.03

Figure 37. Oyster growth on Sandbar substrate (top) and oyster castles (bottom).

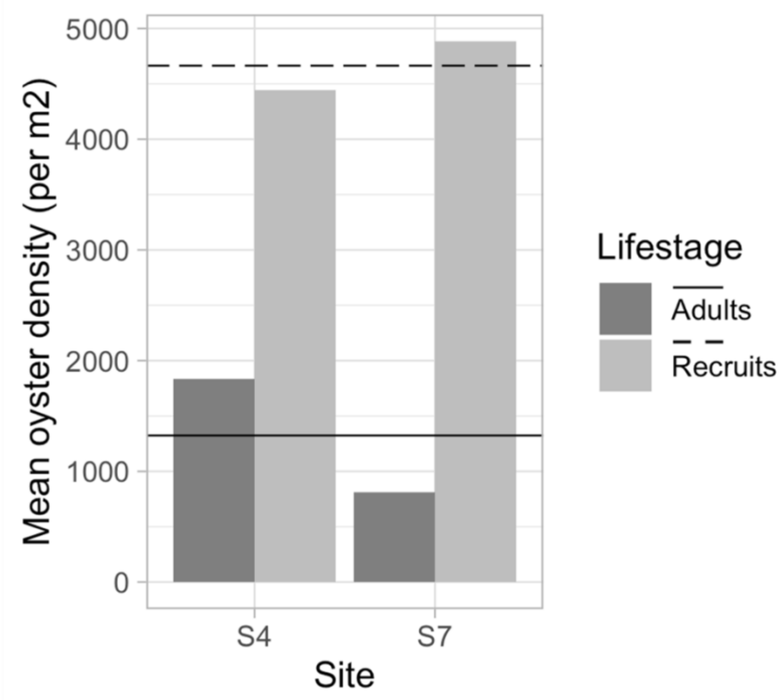

Figure 38. Mean oyster densities sampled in 2021 separated by life stage. Horizontal lines represent overall means (Table 12).

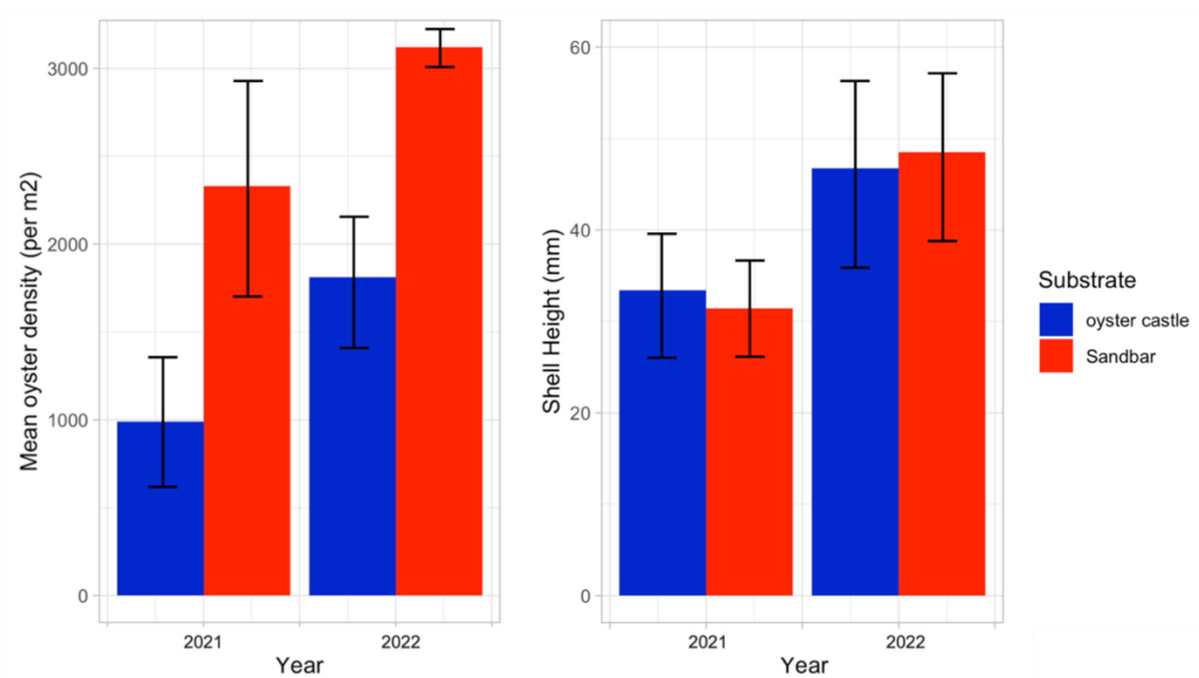

Figure 39. Mean adult oyster densities and shell heights sampled in 2021 and 2022 separated by substrate (Tables 13 & 14). Error bars represent standard deviations around the means. Differences in adult oyster densities are significant between substrates for both sampling years. Differences in adult shell heights are significant between substrates in 2021, but not in 2022. Results from one-way ANOVAs are summarized in Appendix 20.

6. Discussion

6.1 Wave attenuation by oyster reefs

The significant decrease in wave heights observed between the offshore and nearshore wave gauges at S7 when water depths were shallow is indicative of wave attenuation. Similar findings have been reported at other locations with oyster castles in the VCR (Wiberg et al., 2019; Ferguson, 2018; Hogan and Reidenbach, 2021). In shallow water, wave orbital water motion has the ability to interact with the reefs, causing frictional resistance and wave breaking (Wiberg et al, 2019; Hogan and Reidenbach, 2021; Taube, 2013; Piazza et al. 2005). In deeper water, reef structures have less impact on orbital motions resulting in little or no reduction of wave heights. Results from the current study agree with Wiberg et al. (2019) who suggested that fringing oyster reefs will produce little to no decrease in wave energy when water levels are more than a few tenths of a meter above the height of the reef crest. The significant reduction in wave heights for shallow water depths at S7 is not necessarily due to the particular substrate (oyster castles), but rather a result of the presence of any reef structure with a crest elevation high enough to block wave energy.

6.2 Marsh edge morphology

The impact of reef wave attenuation on marsh edges depends on marsh morphology. Marsh islands in Virginia's coastal bays often exhibit a significant edge scarp, with the top of the scarp located between MSL and mean high water (MHW) and a total scarp height of about 1 m above the adjacent tidal flat (McLoughlin et al., 2015). While S7 is gradually sloping with no significant scarp and an edge elevation near MWL, S4 has a pronounced edge scarp that is about 1 m high with an edge elevation closer to MHW. In this study, there was no significant decrease in wave energy due to oyster reef presence at S4 in any water depth. Tonelli et al. (2010) showed that wave attack increases with water depth and that wave thrust, a metric for wave attack acting on marsh edges, is maximum for a vertical scarp and water levels just below the top of the scarp. Since marsh edge erosion is driven by wave attack, which is positively correlated with water depth, it is unlikely that fringing artificial oyster reefs would significantly slow retreat rates for marshes with high edge scarps like the one found at S4. However, marsh edge morphology measurements made in this study show that fringing oyster reefs reduced rates of marsh edge retreat and promoted deposition near the reefs sediments at both S4 and S7. The ability of the reefs at S4 to effectively stabilize the marsh edge is interesting but hard to interpret given the lack of significant wave attenuation evident in the measurements at that site.

A previous study in the VCR that quantified marsh edge morphology using airborne LiDAR data demonstrated the ability of reef-lined marshes to significantly reduce marsh mean and maximum slope values compared to exposed marshes (Hogan et al., 2021). Hogan et al. (2021) hypothesized that oyster reefs facilitate an elongation of the marsh edge by reducing retreat at lower elevations of the marsh edge but not at the top of the scarp, and that these changes in marsh edge morphology are a precursor to changes in marsh retreat. This study found no significant difference in retreat rates between reef-lined and control marshes that ranged from 14 to 79 cm yr^{-1} (Hogan et al., 2021). Other previous studies in the VCR (Taube, 2013; McLoughlin et al., 2015) measured erosion rates along reef-lined and reef-free marshes using Digital Shoreline Analysis (DSAS). Taube (2013) saw average retreat rates of 20 cm yr^{-1} for four reef-lined marshes and McLoughlin et al. (2015) found an average retreat rate of 98 cm yr^{-1} for four reef-free marshes in the VCR. These rates are comparable to the average rates of 53 and 124 cm yr^{-1} for reef-lined and reef-free marshes, respectively, that were measured in the current study. It was not until the current study that changes to marsh retreat at reef-lined and control marsh locations were measured in-situ within the VCR, revealing significantly greater retreat at locations lacking restored reefs.

Overall, results indicate that marsh morphology and elevation are at least as important as the presence of fringing oyster reefs in reducing wave energy driving marsh retreat. The restored reefs in this study are ineffective at attenuating waves unless water depths are near or below reef crest heights. This attenuation is most likely to benefit marshes with edge elevations close to mean sea level. When water depths are low enough, the reefs are blocking waves on the offshore side not allowing for wave energy to propagate over the structures. When water depths are great enough that they can pass over constructed reefs unmodified, wave energy will dissipate either at the marsh edge, tending to drive marsh retreat, or within the marsh platform. The marsh edge appears to be particularly vulnerable in intermediate water depths. Waves reaching the marsh edge when water depths are between the height of the reef crest and the marsh platform (intermediate depths) may be the major driver of marsh edge erosion. Thus, potential benefits of reef-associated wave attenuation are dependent on water depth, marsh edge elevation and morphology (Wiberg et al., 2019; Ferguson, 2018).

Since the largest waves generally accompany high wind and deep-water conditions, constructed oyster reefs with crest elevations below mean sea level are unlikely to be effective at wave attenuation under the highest wave conditions. However, marsh vegetation is effective at attenuating waves even during deeper water conditions (Ferguson, 2018). Ferguson (2018) concluded that combining constructed oyster reefs with vegetated treatments may be an effective and sustainable long-term shoreline stabilization technique in which the reef helps to stabilize the marsh edge while marsh vegetation attenuates wave energy that passes over the reef. Though vegetated treatments were not included in the construction of the living shorelines in the current study, the fringing reefs seem to be effective at reducing edge erosion and promoting local deposition, therefore possibly building up suitable area for natural marsh grass growth and, overtime, increasing wave attenuation by marsh vegetation in deeper water conditions.

6.3 Sediment accumulation

Sediment accumulation tile measurements from this study show some degree of sediment accumulation on the marsh platforms of S4 and S7. However, since marshes naturally accrete vertically by trapping sediment transported onto the marsh during tidal flooding, this accumulation cannot necessarily be attributed to the presence of oyster reefs. One limitation to this portion of the study is that a control site was not used for comparison. Additionally, rainfall can alter the mass of sediment deposited on the tiles.

Marsh platform sediment accumulation measurements do not represent vertical accretion on the tidal flat in and around the constructed reefs. Sediment accumulation data were not collected directly behind the substrates, however photographs revealed buildup in and around the substrate (Figure 30). While grain size samples from this study indicate no wave-driven transport of offshore sediments to nearshore locations, it is clear, at least qualitatively, that the substrates are effective at trapping sediment, even if it is just locally. This suggests the need for a more robust experiment for measuring sediment accumulation, perhaps using a feldspar marker layer to measure vertical accretion through time (Whelan et al., 2016).

6.4 Sediment and infaunal properties

In this study, there were no significant changes to the infaunal community and sediment organic matter composition pre- and post-reef construction. Some oyster restoration studies have seen significant changes to sediment grain size and organic matter content as soon as one year post-restoration (Southwell et al., 2017), while others found that oyster restoration benefits, such as infaunal community enhancement, lagged behind oyster growth (Liu et al., 2018). A study conducted at Short Prong marsh (Hogan and Reidenbach, 2021), a nearby site in the VCR, also revealed no significant difference in sediment organic matter content between dates. However, there were significant changes to the infaunal community over a four year period. Although differences in total infaunal abundance and AFDW were insignificant at sample locations at the Wachapreague sites, worm abundance increased, similar to results at Short Prong marsh (Figures 34 & 35). Infaunal counts in this study were very low when compared to the Short Prong samples, which yielded average counts of 18.8 +/- 2.9, 8.4 +/- 1.8 pre- and one year post-construction (Hogan and Reidenbach, 2021).

Offshore grain sizes were found to be significantly finer than the coarse nearshore grain sizes at both S4 and S7. As the oyster reefs began to grow and mature in 2021, grain sizes at nearshore sites did not significantly change, suggesting that sediments are not moving in from offshore and being trapped in and around the reefs. Rather, the coarser nearshore sediments must be being reworked in and around the Sandbar substrate.

Infaunal communities are subject to large spatial and temporal variability (Grabowski et al., 2005; Ziegler et al., 2017) and change in sediment composition is likely to be a gradual process (Hogan and Reidenbach, 2021). Findings from this study support previous documentation suggesting that different ecosystem services are likely to respond at different timescales (Volaric et al., 2020), with infaunal and sediment composition responses observed more slowly than wave attenuation and oyster growth. A 15-year large-scale experiment in coastal Virginia showed that restored oyster reefs can recover multiple ecological functions and match natural reefs within 6 years (Smith, et. al, 2022). While significant trends were not seen in some metrics in the current study, measurements took place over less than 2 years, suggesting that this site had not yet had the opportunity to reach natural reef functionality. This highlights the need for long-term oyster restoration monitoring.

6.5 Oyster densities and shell heights

Higher and more variable oyster recruit densities during the first post-construction recruitment season suggests that post-settlement processes and site-level differences may be influencing the ability of recruits to grow to maturity (Tedford and Castorani, 2022). Greater adult densities recorded on Sandbar substrate than on oyster castles during both sampling years suggests that Sandbar substrate may provide a more ideal environment for survival to maturity.

Other studies show more successful recruitment and growth on reefs at higher elevations, possibly due to increased flow rates and decreased sedimentation (Hogan and Reidenbach, 2021; Lenihan, 199; Schulte et al., 2009). In this study, Sandbar substrate is at a higher elevation than oyster castles, but still within the range of oyster settlement (Tedford and Castorani, 2022; Hogan and Reidenbach, 2019). It is possible that adult densities are greater on Sandbar substrate than on oyster castles partly due to this elevation difference (Table 11). However, this increase in adult densities could also partly be attributed to more interstitial areas provided by Sandbar substrate that create low shear stresses and allow larvae to remain anchored (Whitman and Reidenbach, 2012; Reidenbach et al., 2009; Crimaldi et al., 2002).

Although greater oyster densities were found at higher elevations on Sandbar substrate, there is probably an elevation threshold above which oyster abundance will decrease, due to increased atmospheric exposure and decreased submergence time (Theuerkauf et al., 2015, Hogan and Reidenbach, 2021). The average Sandbar substrate reef height was -0.2 m NAVD88, placing them about 0.2 m below the maximum elevation in the VCR where oyster reefs have successfully recruited larvae (0 m, NAVD88) (Tedford and Castorani, 2022; Hogan and

Reidenbach, 2019). Since oyster growth adds to elevation with time, there is a need to continuously monitor reef elevation. Additionally, this highlights the ability of natured-based solutions, such as oyster reefs, to adapt and grow to meet changes such as rising sea levels, continuously buffering coastlines.

There are several limitations to this portion of the study, one being that the larval recruitment tiles are not made of the same materials as the artificial reefs. This means that the tiles are not necessarily representative of recruitment on the substrates, Sandbar and oyster castles, themselves. When recruit densities were compared between tiles placed near the substrates and at adjacent bare sites, there was no significant difference, suggesting that the recruit measurements collected on the tiles are indicative of the tiles as substrate and not of the substrate they are placed near. While there were significantly higher adult densities recorded on Sandbar substrate than oyster castles, there was no significant difference in recruit densities between substrates. It might be expected that recruit densities would increase with adult densities. However, this anticipated trend may not be visible due to the differences in substrate materials between the recruitment tiles and the substrates themselves.

Another limitation to this portion of the study includes biological factors, such as algal cover, that could affect oyster restoration success by altering oyster growth and other benthic processes (Thomsen and McGlathery, 2006; Volaric et al., 2019). Although differences in algal cover were not quantified in this study, some algae growth was observed on both types of substrates and on recruitment tiles. Given that Virginia's coastal bays are generally low in nutrient input, especially when compared to nearby areas in the Chesapeake Bay, the interaction between algae and substrate did not seem to significantly limit oyster growth at this location. Additionally, during the first recruitment season, there were large amounts of barnacles that also settled onto the recruitment tiles, making it more difficult to see, count and measure spat. Furthermore, reef complexity makes accurate measurements of oyster densities on both Sandbar substrate and oyster castles more difficult.

While there were higher and more variable recruit densities recorded during the first recruitment season, recruit densities during the second season were relatively low. Since other biological factors, including barnacle settlement and algal growth, also varied between years, it is hypothesized that some external factor, possibly changes to water temperature, pH, salinity, etc., could have been affecting larval recruitment. A study in the Pacific Northwest found that Pacific oyster, *Crassostrea gigas*, larvae are vulnerable to decreased pH, elevated temperature and reduced salinity (Ko et al., 2014). Results from another study indicate that exposure of Eastern oysters to severe ocean acidification in estuarine environments could cause reproductive disruption, therefore decreasing larval settlement and growth (Boulias, M., 2017).

Reproduction of *C. virginica* typically begins when water temperature reaches a daily mean value of 25°C (Galtsoff, 1964). This threshold was reached on May 28, 2021 and May 21, 2022, suggesting earlier warming during the 2022 reproductive season (NOAA Tides and Currents, 2022). In the three week period leading up to the start of the expected spawning seasons, there was up to around 4 inches of rainfall recorded at a nearby meteorological station in Virginia Beach, VA in 2022 and not even 1 inch of rainfall recorded in 2021, implying increased rainwater input and decreased salinity in 2022 (USGS NWIS, 2022). Changes to these types of environmental factors could be affecting oyster larval settlement year to year.

6.6 Study limitations

Field sampling was limited due to constraints of restored reef construction and field logistics. S4 and S7 were the first sites in which reef construction was completed. Other sites, including those with just Sandbar substrate, were not complete until after sampling had already commenced. Additionally, wave data were collected at all sites during all deployment periods. However, due to wave gauge malfunction, not all data were included in analysis. Instead, only deployments with complete records at both the nearshore and offshore locations for each site, S4 and S7, were analyzed.

6.7 Restoration suggestions and future work

Overall, the current reef configurations constructed by TNC at S4 and S7 seem to be effective restoration designs in terms of coastal protection benefits. Oyster substrate at both locations successfully limited marsh edge erosion and fostered larval recruitment and oyster growth, with increased adult densities recorded on Sandbar substrate. Oyster castle treatments successfully attenuated waves in shallow water at S7, a location with a gradually sloping shoreline and no significant scarping where the vertical structures could easily remain grounded. The Sandbar substrate treatment was ideal for the scarped S4 where it was harder to secure oyster castles in place near the marsh scarp. While the oyster castle row offshore at S4 may seem unnecessary, Sandbar substrate in other sections along the marsh with no offshore oyster castle reef were weakened, and in some cases even broke, by wave battering. Therefore, this extra barrier may be important for stabilization and protection of the Sandbar row behind it. In future studies, it might be beneficial to place the same treatments at sites with opposite topography (Sandbar on a gentle slope and oyster castles on a steep scarp). One might also include both treatments at topographically different sites for a clearer comparison of suitability and protection benefits.

A previous study within the VCR at Short Prong Marsh compared several reef configurations with differing widths and heights and concluded increased attenuation with increased reef vertical relief (Hogan and Reidenbach, 2021). The current study used a configuration that was similar to one used for comparison in the Short Prong study containing 2 tiered oyster castles. While the current study saw significant attenuation over the 2 tiered oyster castles in shallow water, the reefs had no effect in intermediate or deeper water. Expanding the height of the reef to match the heights producing the greatest attenuation in the Short Prong study, 4 tiers, might increase marsh protection in the more susceptible intermediate water depths.

With an opportunity to continue monitoring at this site, it may be beneficial to include, at a minimum, measurements of wave attenuation, marsh retreat and oyster densities at least once per year if not more. Significant trends in these metrics have already been demonstrated in the short-term study, so continuing these measurements would only help to create a more robust dataset. A previous study in the VCR showed the ability of oyster reefs to increase wave attenuation through time as they grow and mature, specifically in years 3 and 4 post-construction (Hogan and Reidenbach, 2021). Since this study took place over just two years, this enhanced coastal protection service was not visible, highlighting the need for continued monitoring.

It may also be useful to continue grain size measurements to gain a better understanding of long-term sediment movement. While trends in infaunal and sediment organic matter metrics are likely to be observed more slowly, it is unlikely that the continuation of these measurements would make a significant contribution to this work. A nearby study at Short Prong marsh in the VCR showed no significant trend is sediment organic matter content at a restored reef 3 years post-construction (Hogan and Reidenbach, 2021). Additionally, infaunal counts in this study were very low when compared to the Short Prong pre- and post-construction samples (Hogan and Reidenbach, 2021), suggesting there are not many infauna present at this location to begin with.

6.8 Conclusion

Field measurements from S4 and S7 agree with previous literature that oyster reefs may provide effective and sustainable long-term shoreline stabilization. This study demonstrates that, although restored oyster reefs have limited wave attenuation effectiveness, they are capable of significantly slowing marsh erosion, thus expanding possible area suitable for marsh grass establishment, and fostering larval settlement and oyster growth. Oyster reefs in this study successfully dissipated wind-wave energy, but only when water depths were near or below reef crest heights. In-situ measurements of reef-lined and un-lined marshes showed reefs at this location significantly reduced rates of marsh edge erosion. Additionally, both artificial reef designs in this study successfully fostered larval recruitment and oyster growth, with significantly higher oyster densities recorded on Sandbar substrate than oyster castles. Thus, oyster castles and Sandbar substrate offer a nature-based solution for the coastal protection of the marsh island affronting Wachapreague, VA. Results from this study supplement our understanding of the combination of restored oyster reefs and marshes as a nature-based solution to threats associated with storms and sea level rise.

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8. Appendix

1. 1-way ANOVA test for wave heights by site (4 and 7) in July 2021 (Figure 20, Table 2).

1-way ANOVA			
wave heights by site			
SS	df	F	p
0.07	1	13.64	< 0.01

2. 1-way ANOVA tests for wave heights by location separated by date (Figures 19-22, Table 2).

September 2020 – S4O/S4N

1-way ANOVA			
wave heights by location in shallow water			
SS	df	F	p
0.107	1	92.2	< 0.01

1-way ANOVA			
wave heights by location in intermediate water			
SS	df	F	p
0.142	1	135	< 0.01

1-way ANOVA			
wave heights by location in deep water			
SS	df	F	p
0.0045	1	14.54	< 0.01

September 2020 – S7O/S7N

1-way ANOVA			
wave heights by location in shallow water			
SS	df	F	p
0.252	1	108	< 0.01

1-way ANOVA			
wave heights by location in intermediate water			
SS	df	F	p
0.193	1	49.2	< 0.01

1-way ANOVA			
wave heights by location in deep water			
SS	df	F	p
0.203	1	50.5	< 0.01

July 2021 – S4O/S4M

1-way ANOVA

wave heights by location in shallow water			
SS	df	F	p
0.015	1	4.37	< 0.05

1-way ANOVA

wave heights by location in intermediate water			
SS	df	F	p
0.888	1	153	< 0.01

1-way ANOVA

wave heights by location in deep water			
SS	df	F	p
0.106	1	11.7	< 0.01

July 2021 – S7O/S7N

1-way ANOVA

wave heights by location in shallow water			
SS	df	F	p
0.409	1	175	< 0.01

1-way ANOVA

wave heights by location in intermediate water			
SS	df	F	p
0.077	1	16.1	< 0.01

1-way ANOVA

wave heights by location in deep water			
SS	df	F	p
0.223	1	31.9	< 0.01

November 2021 – S4O/S4M/S4N

1-way ANOVA

wave heights by location in shallow water			
SS	df	F	p
0.057	2	8.45	< 0.01

Tukey multiple comparisons	
S4O/S4N	< 0.01
S4O/S4M	< 0.01
S4M/S4N	> 0.05

1-way ANOVA			
wave heights by location in intermediate water			
SS	df	F	p
0.242	2	34.3	< 0.01

Tukey multiple comparisons	
S4O/S4N	< 0.01
S4O/S4M	< 0.01
S4M/S4N	> 0.05

1-way ANOVA			
wave heights by location in deep water			
SS	df	F	p
0.019	2	7.16	< 0.01

Tukey multiple comparisons	
S4O/S4N	< 0.01
S4O/S4M	< 0.05
S4M/S4N	> 0.05

June 2022 – S7O/S7N

1-way ANOVA			
wave heights by location in shallow water			
SS	df	F	p
0.252	1	108	< 0.01

1-way ANOVA			
wave heights by location in intermediate water			
SS	df	F	p
0.193	1	49.2	< 0.01

1-way ANOVA			
wave heights by location in deep water			
SS	df	F	p
0.203	1	50.5	< 0.01

3. 1-way ANOVA test for marsh platform sediment accumulation by section (4 and 7) (Figure 28, Table 5).

1-way ANOVA			
sediment accumulation by section			
SS	df	F	p
5.93	1	2.686	> 0.05

4. 1-way ANOVA and Tukey multiple comparisons tests for sediment accumulation by location on marsh (edge, middle, interior) (Table 5).

1-way ANOVA			
sediment accumulation by location on marsh			
SS	df	F	p
11.5	2	2.749	> 0.05

Tukey multiple comparisons	
edge-middle	> 0.05
edge-interior	> 0.05
middle-interior	> 0.05

5. 1-way ANOVA test for distance eroded by presence of reefs (Figure 30, Table 6).

1-way ANOVA			
distance eroded by presence of reefs			
SS	df	F	p
121.5	1	10.2	< 0.01

6. 1-way ANOVA test for distance eroded by site (S4 and S7).

1-way ANOVA			
distance eroded by presence of reefs			
SS	df	F	p
7.19	1	1.6	> 0.05

7. 1-way ANOVA test for stem density by section (Figure 31, Table 7).

1-way ANOVA			
stem density by section			
SS	df	F	p
3354	1	5.85	< 0.05

8. 1-way ANOVA test for stem height by section (Figure 31, Table 7).

1-way ANOVA			
stem height by section			
SS	df	F	p
106.9	1	6.925	< 0.05

9. 1-way ANOVA and Tukey multiple comparisons tests for % sediment organic matter content by date (Figure 32, Table 8).

1-way ANOVA			
% sediment organic matter content by date			
SS	df	F	p
59.83	3	2.827	> 0.05

Tukey multiple comparisons	
Oct 2021-Jul 2021	> 0.05
Sep 2020-Jul 2021	> 0.05
Sep 2020-Oct 2021	> 0.05
Jul 2022-Jul 2021	> 0.05
Jul 2022-Sep 2020	> 0.05
Jul 2022-Oct 2021	> 0.05

10. 1-way ANOVA test for % sediment organic matter content by site (S4 and S7).

1-way ANOVA			
% sediment organic matter content by site			
SS	df	F	p
0.038	1	0.04	> 0.05

11. 1-way ANOVA and Tukey multiple comparisons tests for infaunal AFDW by date (Figure 33, Table 9).

1-way ANOVA			
infaunal ash free dry weight by date			
SS	df	F	p
0.017	2	0.14	> 0.05

Tukey multiple comparisons	
Oct 2021-Jul 2021	> 0.05
Sep 2020-Jul 2021	> 0.05
Sep 2020-Oct 2021	> 0.05

12. 1-way ANOVA and Tukey multiple comparisons tests for infaunal AFDW by site (S4 and S7).

1-way ANOVA			
infaunal abundance by date			
SS	df	F	p
0.075	1	1.35	> 0.05

13. 1-way ANOVA and Tukey multiple comparisons tests for infaunal abundance by date (Figure 34, Table 9).

1-way ANOVA			
infaunal abundance by date			
SS	df	F	p
1.73	2	0.925	> 0.05

Tukey multiple comparisons	
Sep 2020	> 0.05
Jul 2021	> 0.05
Oct 2021	> 0.05

14. 1-way ANOVA and Tukey multiple comparisons tests for infaunal abundance by site (S4 and S7).

1-way ANOVA			
infaunal abundance by date			
SS	df	F	p
1.74	1	1.94	> 0.05

15. 1-way ANOVA test for grain size by date (Figure 35, Table 10).

1-way ANOVA			
stem density by section			
SS	df	F	p
2178	1	0.039	> 0.05

16. 1-way ANOVA test for grain size by site (S4 and S7) (Figure 35, Table 10).

1-way ANOVA			
stem density by section			
SS	df	F	p
10225	1	0.186	> 0.05

17. 1-way ANOVA test for grain size by location (nearshore and offshore) (Figure 35, Table 10).

1-way ANOVA

stem density by section

SS	df	F	p
312840	1	68.6	<0.01

18. 1-way ANOVA test for recruit density by substrate for each sampling year (Table 13).

2021

1-way ANOVA

recruit density by substrate

SS	df	F	p
3767520	1	1.89	> 0.05

2022

1-way ANOVA

recruit density by substrate

SS	df	F	p
298148	1	0.59	> 0.05

19. 1-way ANOVA test for adult density by substrate for each sampling year (Figure 39, Table 13).

2021

1-way ANOVA

adult density by substrate

SS	df	F	p
4053511	1	20.7	< 0.01

2022

1-way ANOVA

adult density by substrate

SS	df	F	p
3852060	1	32.5	< 0.001

20. 1-way ANOVA test for adult shell height by substrate for each sampling year (Figure 39, Table 14).

2021

1-way ANOVA

adult shell height by substrate			
SS	df	F	p
846	1	12.24	< 0.001

2022

1-way ANOVA

adult shell height by substrate			
SS	df	F	p
104	1	0.81	> 0.05

21. Wilcoxin signed-rank test for recruit shell height by substrate for each sampling year (Table 14).

2021

Wilcoxin signed-rank test

recruit shell height by substrate	
W	p
19800	< 0.001

2022

Wilcoxin signed-rank test

recruit shell height by substrate	
W	p
3225	< 0.001