Long-term impacts of living shorelines to SAV habitats in Chesapeake Bay

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Abstract

Shoreline erosion rates have been accelerating in many estuaries and coastal embayments, especially in the face of environmental changes such as increased rates of sea-level rise, urbanization, and storminess. Property owners seeking to protect their shorelines are increasingly turning to natural and nature-based features (NNBF) such as living shorelines (defined here as narrow marsh fringes with or without additional structures), which are encouraged or mandated by legislation in many coastal states. Yet, questions remain about their performance over time (i.e., sediment trapping, reducing shoreline erosion, and their impact to adjacent nearshore habitats, especially in areas with submersed aquatic vegetation (SAV)). This study examined these questions at 8, ~10-year-old living shorelines with continuous rock sills and 8 paired reference (unaltered) shorelines. Vegetation surveys were conducted in the created marshes of living shorelines and in the subtidal adjacent to living shorelines and reference shorelines; sediment cores were collected at the same locations. Historical SAV distributions were determined via aerial photographs, and average fetch was measured in GIS. Overall, we found that living shorelines are effective in reducing shoreline erosion and provide additional nutrient storage capacity in the coastal zone. We did not find evidence of negative impacts to adjacent subtidal SAV habitat or distributions from living shoreline installation.

1. Introduction

Like many estuaries and coastal embayments, shorelines in Chesapeake Bay are rapidly eroding, with even more rapid erosion expected in the future due to human and environmental changes (e.g. urbanization, accelerated relative sea-level rise (RSLR) rates, and storms; Church and White 2006; Erdle et al. 2008; Gehrels and Woodworth 2013; Sutton-Grier et al. 2015). Shoreline erosion not only increases sediment input into adjacent waters, degrading water quality, but also results in property loss (Wells et al. 2003). In response, many property owners seek to protect their shorelines with stabilization structures. Past efforts focused on hard structures (e.g. bulkhead, rip rap) that can provide new ecosystem services (e.g. rocky habitat; Seitz et al. 2006) but also can have significant negative ecosystem impacts (Currin et al. 2010 and references therein). As a result, more recent efforts have focused on natural alternatives, including living shorelines.

Living shorelines (LS), narrow marsh fringes with or without additional structures (Burke et al. 2005), are encouraged by legislation in many states, including Maryland. They provide similar ecosystem services as natural marshes (e.g., sediment and nutrient retention, wave attenuation; Currin et al. 2010) but are subject to the same stressors (Davis et al. 2015). Questions remain regarding LS resilience to environmental change, especially in regions like CB where relatively rapid rates of RSLR (Boon 2012) and declining sediment supplies (Weston 2014) have led to widespread marsh loss (Kirwan and Megonigal 2013). Questions also remain regarding the potential long-term impacts of LS to adjacent submersed aquatic vegetation (SAV), which are keystone species in CB (Batiuk et al. 2000). A recent study suggests that LS can decrease fine-sediment inputs from land to adjacent SAV habitats, improving both water-column (water clarity) and substrate (SAV prefer sand; Koch 2001) conditions, but assessing the direct effect of these changes was limited by the absence of SAV at these sites since 1978 and the site-specificity of results (Palinkas et al. 2017).

2. Questions and Hypotheses

This project specifically targeted the questions of long-term performance and impact of LS to SAV habitat and distributions in adjacent shallow waters. We addressed the overarching research questions and linked hypotheses, focusing on LS with continuous rock sills to minimize the potential confounding influence of design.

1) Impacts: how does living shoreline installation impact existing SAV beds?
H1: Living shoreline installation damages or destroys existing SAV beds.

2) Prediction: how can these impacts be predicted?
H2: Living shoreline installation alters the adjacent subtidal SAV habitat.

3) Effectiveness/ecosystem services (question 3): are these impacts "worth it"?
H3a: Sediment and nutrient burial rates will differ between subtidal and intertidal zones, with the highest rates at living shorelines with adjacent SAV.

H3b: Living shoreline installation reduces shoreline-erosion rates, with the lowest rates occurring at sites with SAV after installation.

3. Methods

3.1 Site selection

To minimize variability due to salinity, temperature, large storms, etc., we focused on mesohaline Chesapeake Bay, where shoreline erosion is the dominant sediment source to SAV habitats (Hobbs et al. 1982). To select sites, we produced a weighted overlay of SAV density data from photographs obtained from the Virginia Institute of Marine Science (VIMS) annual aerial surveys; https://www.vims.edu/research/units/programs/sav/index.php)) from 1978 (first available year) to 2005 (target installation year for living shorelines in this study) in ArcGIS, emphasizing recent years and the densest beds (Palinkas and Koch 2012). We overlaid locations of living shoreline construction projects between 2005 and 2008 provided by restoration managers and practitioners (Maryland Department of Natural Resources (MD-DNR), Maryland Department of the Environment (MDE), ShoreRivers, Environmental Concern), as well as Chesapeake Bay Trust, and selected eight sites: four living shorelines with adjacent persistent SAV beds prior to installation and four living shorelines without adjacent SAV prior to installation (Fig. 1, Table 1). Each living shoreline was paired with a nearby (<0.5 km) unaltered shoreline as a reference site.

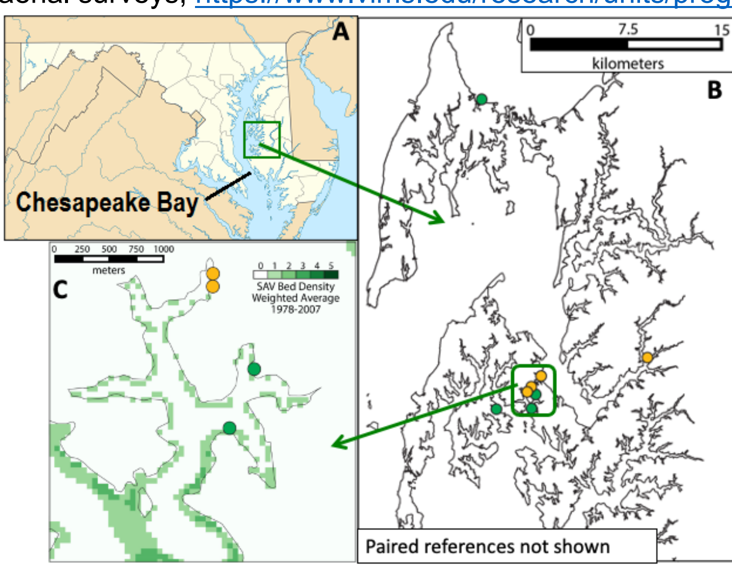

Figure 1. Map of study sites on the eastern shore of Chesapeake Bay in Maryland (A). B) Zoomed in area of the green box in (A); green and yellow circles indicate living shorelines with and without SAV before installation, respectively. C) Zoomed in area of the green box in (B) showing site locations relative to SAV weighted bed density (green shading; darker = more persistent and dense).

3.2 Field methods

Figure 2. Example of transect through intertidal vegetation.

Sediment and vegetation characteristics were assessed in the intertidal, created marsh habitat of the living shoreline and in the adjacent, subtidal shallow-water SAV habitat. At the living shoreline sites, vegetation was assessed along three transects at each site located perpendicular to the shoreline (Fig. 2), extending from the upland edge of the marsh to a maximum of 25 m or 1-m depth offshore. Sampling occurred at 5.0-m intervals along each transect, with sampling intervals shortened to 2.5 meters where necessary to ensure a minimum of 3 sampling points in both the marsh and subtidal zones. At OP, 5 subtidal transects were surveyed, due to the length and complexity of the shoreline. At the paired reference sites, only the subtidal zone was surveyed.

Within the living shoreline marsh, percent cover was estimated at each station using a 0.25 m^2 quadrat marked off in a 10 cm x 10 cm grid (Brower et al. 1998). Marsh canopy height was estimated by measuring the five tallest stems at each station, and all species at the station were identified at least to genus to provide an estimate of species richness for the site. At most sites, stem density was measured at three locations, one each in the high marsh (HM) and low marsh (LM), and one in the transition zone between the two. At EC and SD, all three stem density plots were located in the LM. The low marsh zone was distinguished by the presence of *Spartina alterniflora,* and the transition zone was defined as the landward edge of the *S. alterniflora* range. The HM was characterized by *S. patens* and a variety of other species. Push cores (~20 cm long) were collected in the LM and HM and returned intact to the lab.

In the subtidal zone seaward of the living shoreline and reference shoreline, the presence or absence and species composition of SAV was recorded along each transect. Vibracores (~3 m long) were collected in the subtidal adjacent to the living shoreline and the reference shoreline, as well as push cores to capture relatively undisturbed surface sediments. Cores were returned intact to the lab.

3.3 Laboratory methods

In the lab, sediments were analyzed for grain size, organic matter, and nutrient concentrations. Grain-size analyses were performed by first wet-sieving samples to separate the mud- (<64 μm) and sand-sized (>64 μm) fractions. Then, the sand fraction was dry sieved from 64 μm to 500 μm, using a standard set of 13 sieves, to calculate the median diameter of the sand fraction. Organic content was measured via combustion at 450°C for 4 hours. Particulate nitrogen (N), phosphorous (P), and carbon (C) concentrations were determined for surficial sediments of push cores. Average concentrations were determined for vibracores by analyzing several sections below the surface mixed layer to minimize the effects of enhanced surface concentrations from post-diagenetic mobility of P with iron (Carignan and Flett 1981), biological/physical mixing, and potential interannual variability of surficial concentrations. All analyses were conducted by UMCES's Analytical Services department, which measures N and C concentrations with a CHN analyzer (Cornwell et al. 1996) and P concentrations via ashing/colorimetry (Aspila et al. 1976).

Sedimentation rates were determined from the naturally occurring radioisotopes ^{7}Be (push cores; half-life 53.3 days) and ^{210}Pb (vibracores; half-life 22.3 years), which capture seasonal- and decadal-scale processes, respectively. Both have been used previously in Chesapeake Bay SAV and marsh habitats (Palinkas and Koch 2012; Palinkas et al. 2013; Palinkas and Engelhardt 2016; Russ and Palinkas 2018). ^{7}Be is produced in the atmosphere and is delivered by precipitation to land, where it adsorbs onto sediments that are subsequently eroded and transported into adjacent waters (Olsen et al. 1986). Bulk sediment from the topmost 1 cm of each push core was analyzed for its ^{7}Be activity, using gamma spectroscopy and following (Palinkas et al. 2013). For each core, analysis proceeded with every 1-cm section down the core until ^{7}Be activity was not detected; one additional section below this horizon was counted. This analysis yields mass deposition rates (g/cm^2/y) that can be translate to linear deposition rates (cm/y) via multiplication by the bulk density. ^{210}Pb is supplied to sediments by precipitation, runoff, and decay of its effective parent ^{226}Ra (Nittrouer et al. 1979). ^{210}Pb activities will be measured via alpha spectroscopy, following (Palinkas and Engelhardt 2016). Measured activities of both ^{7}Be and ^{210}Pb were decay-corrected to the time of collection and normalized to the corresponding mud content, since they preferentially absorb onto fine particles (Andersen et al. 2011).

Sediment ages were calculated from depth-integrated ^{210}Pb inventories using the Constant Initial Concentration (CIC) model, which allows for time-varying sedimentation (Appleby and Oldfield 1978). The age (t) of sediments at depth z is:

$$t = \left(\frac{1}{\lambda}\right) ln\left(\frac{A_o}{A_z}\right)$$

where λ is the decay constant of ^{210}Pb (0.031 y^{-1}), A_z is the total inventory of excess ^{210}Pb activity beneath depth z, and A_0 is the total inventory of excess ^{210}Pb activity in the entire core. Sediment ages were used to identify horizons in down-core profiles corresponding to years of living shoreline installation. “Post-installation” sediments reside above these horizons; “pre-installation” sediments are defined as the portion of cores below these horizons that represent the equivalent time period, which minimizes inclusion of historical changes present at the base of cores.

3.4 Ancillary data (erosion rates, fetch, SAV distributions)

Historical shoreline-erosion rates were calculated from the Maryland Coastal Atlas (MCA; https://dnr.maryland.gov/ccs/coastalatlas/Pages/default.aspx), which contains data from digitized shorelines between 1841 and 1995 (Hennessee et al. 2002; Hennessee et al. 2003). To calculate historical erosion rates, we imported digitized shorelines from 1942 and 1994 into ArcGIS, measured the difference at the shoreline perpendicular to the subtidal coring site (for consistency between living shorelines and natural shorelines), then divided by the time elapsed. To calculate current erosion rates, we obtained georeferenced aerial photographs from VIMS (JJ Orth and Dave Wilcox; pers comm) taken in 2003 for the pre-installation shoreline and 2017 for the post-installation shoreline (corresponds to our first field survey); shorelines were digitized and measured at identical points as before. Because of the mixed methods, shoreline-change rates between 1994 and 2003 could not be determined. Additional data on land use and bank height were obtained from the Center for Coastal Resource Management (http://ccrm.vims.edu/gis_data_maps/shoreline_inventories/) via the MD-DNR Coastal Atlas. The survey date was 2004, so these represent pre-installation conditions.

Fetch distance is often used as a proxy for physical energy, with a longer fetch implying higher wave exposure (Keddy 1982). Fetch was quantified by measuring the distance to land along 40 vectors radiating from each site in ArcGIS and then averaging all distances, including zeroes (Koch et al. 2006; Palinkas et al. 2016). This yields the average fetch and does not emphasize the dominant direction of strongest winds. The wave climate of Chesapeake Bay is both fetch- and depth-limited (Hardaway 1995), and so the fetch exposure and water depth, which is similar among study sites, provides an estimate of relative wave energy among sites.

SAV distributions were determined from the VIMS Interactive SAV Map (https://www.vims.edu/research/units/programs/sav/access/maps/index.php) from 1997 (10 years before first field survey) to 2019, except for 2018 when areas around several sites were not mapped. Estimated bed density was obtained for the area immediately adjacent to all sites (1-4, with 1 being very sparse and 4 being dense) for these years. To separate structure effects on SAV distribution from interannual fluctuations in regional SAV distribution, the time series of SAV bed density for each site was compared with the time series of SAV area (hectares) for the broader region, defined here as the USGS 7.5-minute quadrangle ("quad"), in which the site was located.

3.5 Statistics and data synthesis

Statistical analyses were conducted using R or SigmaPlot statistical software. Subtidal vegetation data are reported as the mean (n=3) percent vegetated sites per transect. Data from the paired LS and reference sites were tested for significant differences (p=0.05) using t-tests, or the Mann-Whitney Wilcoxon test if the normality requirement was not met. T-tests were also used to compare pre- and post-installation sediment conditions. ANOVA was used to compare sediment characteristics among marsh, subtidal adjacent to living shorelines, and subtidal adjacent to reference shorelines. It is important to note current discussions on statistics and "significance", since p values do not measure the importance of results (Wasserstein and Lazar 2016) and should not be the sole basis for management decisions (Smith 2020). Indeed, geological field data often preclude robust statistical analysis (Krumbein 1960), and the number of stations and samples in this study were relatively low. Thus, for sediment analyses, we allowed a higher p value of 0.10 to indicate "significance" to identify differences that may be physically meaningful.

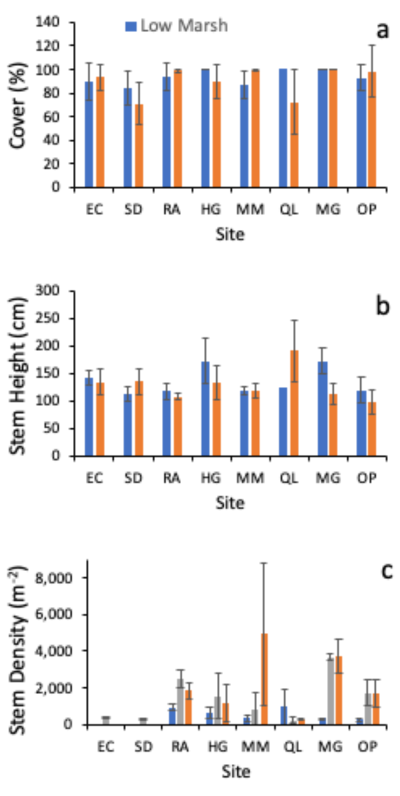

Figure 3. Intertidal vegetation characteristics in living shorelines: (a) mean percent cover and (b) mean stem height in HM and LM, and (c) mean stem density in the HM, LM, and the transition zone between them.

4. Results

4.1 Vegetation observations

4.1.1 Field observations

Vegetation survey dates, transect lengths and dominant species for each site are summarized in Table 2. *Spartina alterniflora* (cordgrass) was the dominant LM species at most sites, with *Phragmites australis* (phragmites) dominant along two transects at site QL. Typically, the HM was more diverse, with *S. patens* (saltmarsh hay) dominant at a majority of sites, but other species were co-dominant at some sites.

In the marsh component of the living shorelines, mean percent cover was fairly uniform, ranging from ~75-100% at all sites, with little difference between HM and LM or between sites (Fig. 3a). Stem height was similarly uniform, ranging between 125-175 cm (Fig. 3b). Although the stems of some species found in the high marsh were decumbent or prostrate (e.g. *S. patens*), their length was similar to the upright stems of *S. alterniflora* in the LM. In addition, taller species such as Phragmites and *Iva frutescens* (high tide bush) were sometimes found in the HM, particularly at the landward edge, resulting in high mean stem height at some locations. Unlike percent cover and stem height, stem densities varied with location within the marshes (Fig. 3c). Lower stem densities were associated with *S. alterniflora* in the LM, while higher densities were associated with *S. patens* and *Distichlis spicata,* which have finer stems than *S. alterniflora*, in the transition and HM zones.

In the subtidal zone adjacent to the living shorelines, SAV was present at 6 of the 8 sites surveyed (Fig. 4). *Ruppia maritima* (widgeon grass) was the dominant subtidal species found at all sites where SAV was observed. It was found exclusively at most sites, along with *Zannichellia palustris* (horned pondweed) at sites EC, HG and HGC. No SAV was found at QL, QLC, MG, MGC, or MMC, and just one occurrence of *Ruppia* was found at MM. While there was variability in the percent vegetated sites between the living shorelines and paired reference shorelines, there were no significant differences (p=0.05) among the pairs.

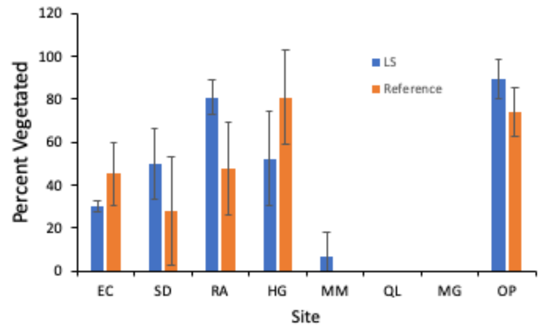

Figure 4. Mean (n=3) percent vegetated sampling stations per subtidal transect at living shoreline (LS) and paired reference sites.

4.1.2 Historical SAV data

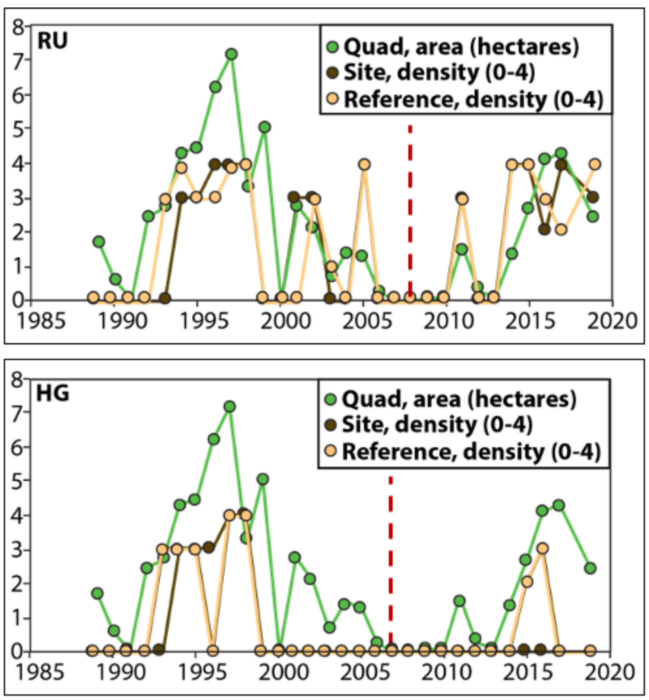

Figure 5. Historical SAV data for RU (top) and HG (bottom). Brown and yellow circles show density classes at the living shoreline and reference shoreline, respectively, and green circles indicate SAV area in the region (quad), which is the same for both sites. The dashed red line marks the installation year of the living shoreline.

Regional SAV distributions varied considerably over time. SAV area in the quad containing most of our sites was low to non-existent at the beginning of the record, increasing to its peak in 1997, followed by a decline to a period of low to non-existent in the mid-late 2000s, rebounding to the secondary peak 2017 (Fig. 5). This record provides background needed to interpret SAV variability at study sites, indicating whether the general area is conducive to SAV; e.g. suitable water quality and/or seed availability. When the time series of SAV bed density at individual sites was plotted along with the regional trend, SAV at most sites followed the regional trend, both before and after installation (Fig. 5, top, showing trends at RU as an example), with three exceptions. At HG (Fig. 5, bottom same quad as RU), SAV disappeared in 1999, when SAV area in the quad also disappeared, but SAV did not rebound at HG (as of 2019) like it did in the quad. (While we found SAV at HG in 2018, the area was not completely mapped via aerial photography that year.) SAV at the reference site also followed this pattern, except for 2015 and 2016, so it is unlikely that the lack of SAV is related to the living shoreline installation. SAV also disappeared at QL and its reference shoreline after the installation year. MG and its reference site MGC did not have SAV at any point in the record even though the quad did have occasional SAV.

4.2 Sediment observations

4.2.1 Surface sediment characteristics

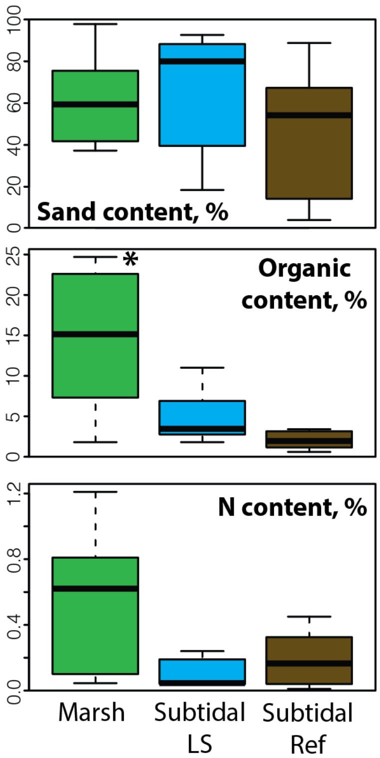

Figure 6. Characteristics of surface sediments in the three environments. Average characteristics were similar, except the organic content was much higher in the marshes.

Characteristics of surface sediments varied widely within each environment (Table 3). In the created marshes of the living shorelines, sand content ranged from 37.3% at RU to 97.8 at MM, averaging 61.1±21.9%. Organic content ranged from 1.8% at MM to 24.7% at SD, averaging 14.6±8.6%. PC ranged from 0.53% at MM to 15.38% at MG, averaging 6.62±5.67%. PN ranged from 0.05% at MM to 1.21% at MG, averaging 0.57±0.45%. PP ranged from 0.05% at MM to 4.72% at EC, averaging 1.56±1.82%. Sand percent was correlated with organic content ($p=0.009$) and nutrient content ($p=0.09$, $p=0.07$, $p=0.06$ for C, N, and P, respectively). In the subtidal adjacent to living shorelines, sediment was also generally sandy but with wider range from 18.4% at EC to 92.6% at MG, averaging 65.8±29.9%. Organic content ranged from 1.8% at OP to 11.0 at RU, averaging 4.9±3.2%. PC ranged from 0.26% at OP to 8.9 at QL, averaging 1.78±2.96%. PN ranged from 0.04% at multiple sites (OP, RU, MG, MM) to 0.8% at QL, averaging 0.17±0.26%. PP ranged from 0.11% at MG to 0.85% at EC, averaging 0.43±0.26%. Sand was correlated with PP ($p=0.008$) but no other sediment parameters. Lastly, in the subtidal adjacent to reference shorelines, sediment was muddier, with sand contents ranging from 4.1% at HG to 88.8% at MG and averaging 45.5±31.0%. Organic content ranged from 0.6% at OP to 8.3% at RU, averaging 2.7±2.5%. PC ranged from 0.13% at OP and MM to 8.9% at QL, averaging 2.41±2.99%. PN ranged from 0.01 at MM to 0.82% at QL, averaging 0.24±0.27%. PP ranged from 0.04% at MM to 1.29% at EC. Sand content was correlated with PP percent ($p<0.001$) but no other sediment parameters.

Sand content was not significantly different among the three environments (Figure 6). Organic content in the marshes was significantly higher than in the subtidal adjacent to living shorelines ($p=0.005$) and reference shorelines ($p<0.001$), but organic content in the subtidal did not differ according to shoreline type. PC and PN was significantly different only between marshes and subtidal adjacent to living shorelines ($p=0.08$ and $p=0.09$, respectively). No differences were found for PP.

4.2.2 Seasonal-scale sedimentation rates

Seasonal (^{7}Be-derived) mass deposition rates were relatively high in all three environments. In the marshes, rates ranged from 0.29 g/cm^2/y at MG to 2.43 g/cm^2/y at QL, averaging 0.75±0.70 g/cm^2/y. This translates (via multiplication by bulk density) to an average linear rate of 1.21±0.85 cm/y, which is well above the rate of RSLR at Cambridge since 2005 (0.56 cm/y). Five of the sites have linear rates above RSLR (QL, RU, HG, SD, EC), and three have

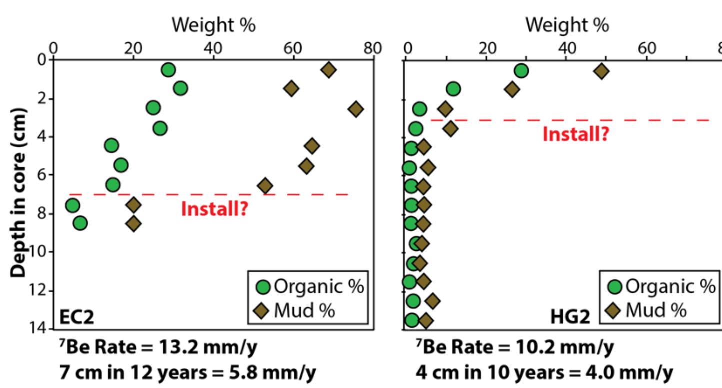

Figure 7. Profiles of mud and organic content in cores at two marsh sites.

rates below RSLR (OP, MG, MM). It is important to keep in mind that the time scales represented by ^{7}Be measurements (seasonal) are much shorter than RSLR (decadal), so caution should be used in making direct comparisons of rates. Our cores were collected during the growing season, when plants can enhance sedimentation via trapping and/or organic matter production, and so our rates should be considered upper estimates. Annual-scale deposition rates integrate periods of non-deposition/erosion in the winter, when plants are absent and/or physical conditions are more energetic. Annual rates can be estimated by examining down-core changes in sediment characteristics, which allow insight into how living shorelines evolve post-installation (Fig. 7). There is a clear distinction between the sandy sediment laid down at installation (bottom of cores) and the increase in mud and organic content that presumably arises from trapping and/or production as plants mature. Linear deposition rates can be calculated by assuming the change in sediment characteristics corresponds with installation. These rates are typically much lower than the ^{7}Be-derived linear rates as they reflect averages over the lifetime of the marsh.

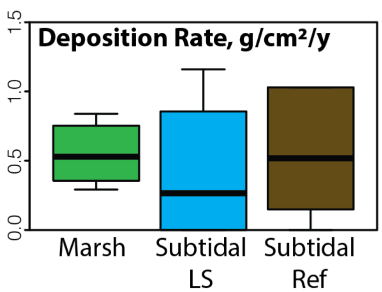

Figure 8. Mass deposition rates in the three environments.

Nonetheless, we emphasize the ^{7}Be-derived mass deposition rates in the marshes to be consistent with measurements in the subtidal, where down-core changes in sediment texture are not as apparent. In the subtidal adjacent to living shorelines, deposition rates range from 0 (^{7}Be was not detected) at EC to 1.16 g/cm$^{2/}$y at QL, averaging 0.42±0.49 g/cm$^{2/}$y, which translates to an average linear deposition rate of 0.87±1.05 cm/y. In the subtidal adjacent to reference shorelines, mass deposition rates ranged from 0 (not detected) at RUC to 2.92 g/cm$^{2/}$y at QL, averaging 0.86±1.07 g/cm$^{2/}$y or 1.31±1.37 cm/y. Mass deposition rates were not significantly different among the three environments (Fig. 8).

4.2.3 Role of vegetation and nutrient burial rates

There were no significant differences in subtidal sediment characteristics or deposition rates when sites were grouped as having SAV present versus absent before installation. While the sample size was low (n=4 in each group), and the variability was quite high, this finding is confounded by the fact that we observed SAV at 2 sites in the "SAV absent" group (EC, SD), and we did not observe SAV at a site in the "SAV present" group (QL, which had very high sedimentation rates in all three environments). More sites are needed to tease apart differences due to site variability versus differences due to plant presence/absence. Results in the marsh are clearer – mass deposition rates, except for QL, were correlated with average stem density (Fig. 9).

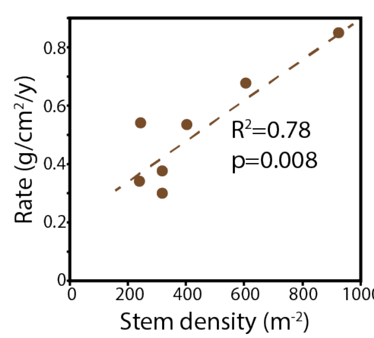

Figure 9. Marsh deposition rates versus stem density, excluding QL (rate 2.43 g/cm²/y, stem density 190 m⁻²).

Nutrient burial rates are given in Table 3c, including a breakdown of rates if subtidal sites are grouped by those with or without pre-existing SAV for reference. Results in the subtidal were highly influenced by site QL, which had anomalously high deposition rates and nutrient concentrations (although nutrients were not analyzed for the marsh sample). Including QL, there were no significant differences in PC, PN, or PP burial rates among the three environments. Excluding QL, marsh PC and PN rates were significantly higher than in the subtidal adjacent to living shorelines (p=0.07 and p=0.08, respectively) but not reference shorelines, and there were no differences in PP. The total nutrient storage in the coastal zone (intertidal plus subtidal components) is much higher at living shorelines because of the additional storage in marshes, which can account for >90% of the total. More sites are needed so that statistics are robust enough to produce reliable total numbers.

4.2.4 Changes in subtidal sediments pre- versus post-installation

As described in the methods, ^{210}Pb geochronology was used to determine the depth horizon in subtidal vibracores corresponding to living shoreline installation. Characteristics of sediments above this horizon were averaged to obtain "post-installation" conditions. Characteristics of sediments for the equivalent time span below this horizon were averaged to obtain "pre-installation" conditions. The same time span was used for the living shoreline and its corresponding reference shoreline.

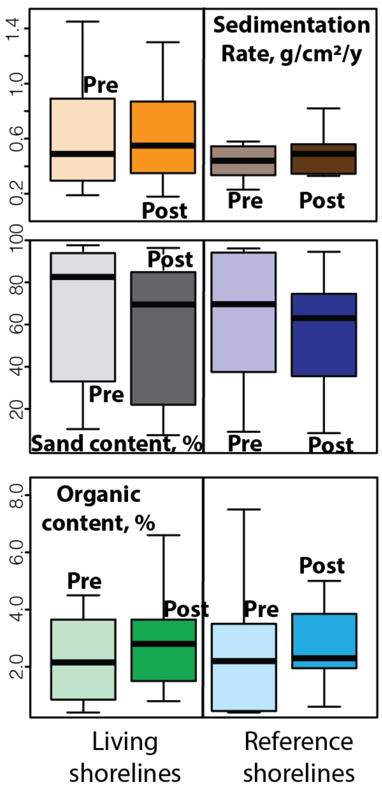

Figure 10. Pre- and post-installation sediment characteristics, showing no significant differences among groups.

There were no significant differences in pre- versus post-installation mass sedimentation rates (p=0.98), sand content (p=0.11), or organic content (p=0.11) at the living shorelines. There were also no significant differences in pre- versus post-installation mass sedimentation rates (p=0.73), sand content (p=0.25), or organic content (p=0.11) at the reference shorelines. All parameters were similar for living shorelines and reference shorelines, both before and after installation. The variability within each group is much larger than differences among groups (Fig. 10). Thus, there is no clear systematic change associated with living shoreline installation.

However, there were significant changes at some individual sites (Table 4). For example, mud content at MG increased significantly after installation at both the living shoreline and the reference shoreline. Trends are not always the same at living shorelines and corresponding reference shorelines (e.g. the sedimentation rate at OP did not change after installation, but it decreased at the reference site), highlighting the site-specificity of sediment changes. It may also reflect the low number of sites and inherent within-site variability rather than change over time.

4.3 Ancillary data (land use, fetch, shoreline-change rates)

Land use differed between shoreline types, with living shorelines mostly being associated with residential areas, and reference shorelines being mostly forested or scrub-shrub areas, most likely reflecting intentional decisions by property owners. Fetch was similar for living shorelines and reference shorelines (p=0.48). Fetch was correlated with mass deposition rates in the marshes only when the apparent outlier at QL (high deposition rate and high fetch) was included. Deposition rates in the subtidal adjacent to living shorelines were correlated with fetch, whether QL was included (p=0.03) or not (p=0.07). In the subtidal adjacent to reference shorelines, deposition rates were correlated with fetch (p=0.06, with or without QL).

We were unable to calculate shoreline-change rates for MG, because data from historical surveys and aerial photographs were not available. We were able to calculate a historical rate for the MG living shoreline site but not the reference site, and so we excluded MG from this discussion. Note that negative rates indicate erosion (shoreline moved landward) and positive change rates indicate accretion (shoreline moved seaward).

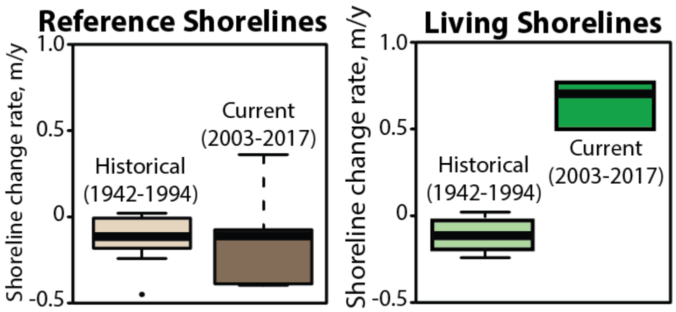

Figure 11. Shoreline-change rates at reference shorelines (left) and living shorelines (right). Negative rates indicate erosion; positive rates indicate accretion.

At reference sites, historical (1942-1994) shoreline-change rates ranged from -0.45 m/y at MM to no change (0 m/y) at SD, with an average of -0.14±0.16 m/y. Current (2003-2017) shoreline-change rates ranged from -0.40 m/y at QL to +0.50 m/y at SD, with an average of -0.06±0.46 m/y. There was no statistical difference between historical and current shoreline-change rates at the reference sites (p=0.83) (Fig. 11).

At living shoreline sites, historical shoreline-change rates ranged from -0.25 m/y at RU to +0.02 m/y at HG, with an average of -0.11±0.11 m/y. Current shoreline-change rates ranged from +0.02 m/y at QL to +0.78 m/y at HG, with an average of +0.46±0.30 m/y. There was a significant difference in change rates (p<0.001) (Fig. 11). It is important to note that the current change rate reflects an instantaneous change associated with installation rather than an actual rate, given that installation practices typically build the shoreline seaward (Fig. 12). Historical shoreline-change rates were not significantly different between living shorelines and references, but current shoreline-change rates were much higher at living shorelines (p=0.003).

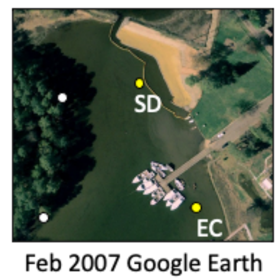

Figure 12. Google Map image showing the shoreline protrusion at SD during installation, and EC already installed. Corresponding reference sites are shown by the white circles.

Historical shoreline changes at sites with SAV were similar to those without SAV (p=0.70, all sites together). Similarly, no differences in current shoreline-change rates were found at reference shorelines with versus without SAV (p=0.77), or at living shorelines with versus without SAV (p=0.84).

Summary

The results detailed above can be integrated evaluate the hypotheses.

H1) Living shoreline installation damages or destroys existing SAV beds.

This hypothesis was **not supported**. There was no obvious difference in SAV distributions beyond the installation footprint before versus after living shorelines were installed. Instead, SAV distributions adjacent to all shorelines (living or reference) generally followed regional trends, except where it disappeared at both the living shoreline and its corresponding reference shoreline.

H2: Living shoreline installation alters the adjacent subtidal SAV habitat.

This hypothesis was **not supported.** When viewed as an ensemble, there were no significant differences in subtidal sediment characteristics before versus after installation of living shorelines, either at the sites themselves or for the equivalent time period at reference shorelines. Changes at some individual sites were significant, but there was no indication of systematic change linked to living shoreline installation. Sediment changes did not appear to influence SAV distributions (see H1).

H3a: Sediment and nutrient burial rates will differ in subtidal and intertidal zones.
This hypothesis was **supported**. On average, sedimentation rates were similar between intertidal and tidal zones. PC and PN burial rates were significantly higher in marshes than in the subtidal adjacent to living shorelines but not reference shorelines; there were no differences in PP. The total nutrient storage in the coastal zone (intertidal plus subtidal components) is much higher at living shorelines because of the additional storage in marshes, which can account for >90% of the total. More sites are needed so that statistics are robust enough to produce reliable total numbers.

H3b: Living shoreline installation reduces shoreline-erosion rates, with the lowest rates occurring at sites with SAV after installation.
This hypothesis was **partially supported.** While historical rates of shoreline erosion persisted at reference shorelines, shorelines accreted (moved seaward) at living shorelines due to installation. However, the presence of SAV did not significantly influence shoreline-change rates.

Overall, we find that living shorelines are effective in reducing shoreline erosion and provide additional nutrient storage capacity in the coastal zone. We did not find evidence of negative impacts to adjacent subtidal SAV habitat or distributions from living shoreline installation.

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Table 1. Name, location, install year, length, pre-existing SAV status (yes = persistent, dense; no = absent), and average fetch for each living shoreline site (first 8 rows), along with 2004 riparian land use. All sites had bank heights of 0-5 ft in 2004. The second 8 rows give information for the corresponding reference site, omitting install year and length

Name	Location N latitude; W longitude	Install Year	Length ft (m)	Pre-existing SAV	Fetch (m)	Riparian land use
Queens Landing (QL)	38° 58.944' 76° 16.757'	2005	600 (182.9)	Yes	920.1±2596.8	Residential
Oppenheim (OP)	38° 46.335 76° 15.658	2006	440 (134.1)	Yes	436.2±811.6	Residential
Ruesch (RU)	38° 46.092 76° 13.389	2008	1330 (405.4)	Yes	246.4±243.9	Residential
Hatton Garden (HG)	38° 46.459 76° 13.299	2007	1860 (566.9)	Yes	93.8±121.8	Forest
San Domingo (SD)	38° 46.852 76° 13.593	2007	770 (234.7)	No	76.2±88.2	Residential
Environmental Concern (EC)	38° 46.803 76° 13.584	2005	550 (167.6)	No	97.6±124.0	Residential
Myrtle Grove (MG)	38° 48.164 76° 7.247	2004	1500 (457.2)	No	231.7±287.3	Grass
Maritime Museum (MM)	38° 47.352 76° 13.258	2008	615 (187.5)	No	507.6±1143.2	Commercial
QL reference	38° 58.826 76° 16.753			Yes	960.2±2629.7	Scrub-shrub
OP reference	38° 46.274 76° 15.569			Yes	504.2±785.6	Forest
RU reference	38° 45.848 76° 13.175			Yes	146.7±195.6	Residential
HG reference	38° 46.262 76° 13.038			Yes	91.3±99.9	Forest
SD reference	38° 46.852 76° 13.630			No	56.8±57.1	Forest
EC reference	38° 46.810 76° 13.646			No	65.2±75.0	Forest
MG reference	38° 48.476 76° 7.498			No	206.9±336.3	Scrub-shrub
MM reference	38° 47.938 76° 11.754			No	877.7±1393.0	Forest

Table 2. Summary of vegetation surveys of living shoreline and paired reference sites. Species are *S. alterniflora* (*S. alt.*), *S. patens* (*S. pat.*). *Bulboschoenus robustus* (*B. rob.*), *P. australis* (*Phrag.*), *I. frutescens* (*Iva*) and *R. maritima* (*Ruppia*). Where dominant SAV species is not shown, no SAV was observed.

Site	Marsh Survey Date	Mean Marsh Transect Length (m)	Marsh Species Richnes	Dominant Marsh Species (LM/HM)	Subtidal Survey Date (LS& Ref.)	Mean LS Subtidal Transect Length (m)	Dominant SAV Species	Mean Reference Subtidal Transect Length (m)	Dominant Reference Plant Species
EC	9/29/17	12.5	28	S. alt./S. pat., B. rob.	7/5/18	22.5	Ruppia	25.0	Ruppia
SD	9/29/17	11.7	17	S. alt./S. pat.	7/5/18	25.0	Ruppia	25.0	Ruppia
RU	10/18/17	10.0	11	S. alt./S. pat.	7/11/18	25.0	Ruppia	25.0	Ruppia
HG	10/18/17	18.0	17	Phrag., S. alt./S. pat.	7/11/18	25.0	Ruppia	25.0	Ruppia
MM	7/26/18	8.0	8	S. alt./S. pat.	7/18/18	9.7	Ruppia	15.8	
QL	10/2/18	4.8	15	Phrag., S. alt./Phrag.	8/15/18	25.0		25.0	
MG	10/3/18	20.0	15	S. alt./ S. pat.	8/29/18	25.0		25.0	
OP	10/19/18	25.0	15	S. alt./ S. pat., Iva	8/8/18	25.0	Ruppia	25.0	Ruppia

Table 3a. Characteristics of surface sediments and seasonal-scale sediment deposition rates in the created marshes of living shorelines. Mass deposition rates are listed in the top row of the cell, and corresponding linear rates are listed in parentheses in the bottom row of the cell.

Site	**Sand%**	**Organic%**	**PC%**	**PN%**	**PP%**	**Deposition rate, $g/cm^2/y$ (cm/y)**
Queens Landing (QL)	70.0	8.9	NA	NA	NA	2.43 (3.04)
Oppenheim (OP)	81.0	5.72	4.75	0.45	0.40	0.39 (0.46)
Ruesch (RU)	37.3	13.6	8.80	0.79	2.75	0.84 (1.17)
Hatton Garden (HG)	69.2	16.7	1.00	0.10	0.62	0.67 (1.02)
San Domingo (SD)	45.4	24.7	NA	NA	NA	0.53 (1.56)
Environmental Concern (EC)	38.1	23.8	9.28	0.81	4.72	0.53 (1.32)
Myrtle Grove (MG)	49.6	21.4	15.38	1.21	0.80	0.29 (0.29)
Maritime Museum (MM)	97.8	1.8	0.53	0.05	0.05	0.37 (0.42)
Average	61.1±21.9	14.6±8.6	6.62±5.67	0.57±0.45	1.56±1.82	0.75±0.70 (1.21±0.85)

Table 3b. Characteristics of surface sediments and seasonal-scale sediment deposition rates in the subtidal, shallow waters adjacent to living shorelines (top 8 rows) and corresponding reference shorelines (bottom 8 rows). Mass deposition rates are listed in the top row of the cell, and corresponding linear rates are listed in parentheses in the bottom row of the cell.

Name	Sand%	Organic%	PC%	PN%	PP%	Deposition rate, $g/cm^2/y$ (cm/y)
Queens Landing (QL)	85.4	3.9	8.9	0.80	0.49	1.16 (2.63)
Oppenheim (OP)	79.8	1.8	0.26	0.04	0.16	NA
Ruesch (RU)	80.1	11.0	0.32	0.04	0.47	0.42 (0.53)
Hatton Garden (HG)	55.4	5.5	0.43	0.05	0.38	0 (not detected)
San Domingo (SD)	23.7	8.3	2.14	0.24	0.73	0.11 (0.44)
Environmental Concern (EC)	18.4	3.0	1.5	0.14	0.85	0 (not detected)
Myrtle Grove (MG)	92.6	2.6	0.34	0.04	0.11	0.85 (1.63)
Maritime Museum (MM)	91.0	2.9	0.32	0.04	0.31	NA
Average	65.8±29.9	4.9±3.2	1.78±2.96	0.17±0.26	0.43±0.26	0.42±0.49 (0.87±1.05)
QL reference	53.2	2.3	8.9	0.82	0.57	2.92 (3.80)
OP reference	73.6	0.6	0.13	0.02	0.17	NA
RU reference	60.9	8.3	0.44	0.06	0.14	0 (not detected)
HG reference	4.1	3.4	1.57	0.18	1.04	0.15 (0.44)
SD reference	13.0	1.3	2.14	0.24	0.73	0.51 (1.83)
EC reference	15.4	1.0	1.52	0.15	1.29	0.53 (0.62)
MG reference	55.2	2.9	4.49	0.45	0.5	1.03 (1.18)
MM reference	88.8	1.6	0.13	0.01	0.04	NA
Average	45.5±31.0	2.7±2.5	2.41±2.99	0.24±0.27	0.66±0.56	0.86±1.07 (1.31±1.37)

Table 3c. Average nutrient burial rates, calculated from values in tables above. The top row in cells for the subtidal includes all sites; the bottom row excludes QL, because it was a clear outlier in both sedimentation rate and nutrient concentrations.

	C burial, $mg/cm^2/y$	**N burial, $mg/cm^2/y$**	**P burial, $mg/cm^2/y$**
Marsh	32.0±28.3	2.8±2.5	9.3±11.4
Subtidal, living shorelines	54.5±102 (1.3±1.3)	1.7±3.7 (0.16±0.15)	6.3±5.9 (0.74±0.82)
Subtidal, reference shorelines	18.3±41.6 (13.4±18.8)	5.1±9.4 (1.3±1.9)	1.6±2.2 (4.2±3.3)

Nutrient burial rates for sites grouped by pre-existing SAV presence or absence. As above, the top row in cells for the subtidal with SAV includes all sites; the bottom row excludes QL.

C burial rates, $mg/cm^2/y$

	SAV present	**SAV absent**
Marsh	32.2±36.3	31.9±26.0
Subtidal, living shorelines	34.9±59.2 (0.67±0.95)	1.8±1.5
Subtidal, reference shorelines	87.5±150 (1.2±1.7)	21.6±21.4

N burial rates, $mg/cm^2/y$

	SAV present	**SAV absent**
Marsh	2.9±3.2	2.7±2.2
Subtidal, living shorelines	3.2±5.3 (0.13±0.19)	0.20±0.18
Subtidal, reference shorelines	8.1±13.7 (0.08±0.11)	2.2±2.2

P burial rates, $mg/cm^2/y$

	SAV present	**SAV absent**
Marsh	9.5±11.8	9.1±13.7
Subtidal, living shorelines	2.6±2.9 (0.99±1.4)	0.58±0.51
Subtidal, reference shorelines	6.1±9.2 (0.78±1.1)	6.5±1.3

Table 4a. Pre- and post-installation sedimentation rates in the subtidal adjacent to living shorelines and corresponding reference shorelines. Data are averaged over the equivalent time span for both. For example, QL was installed in 2005, so post-installation data represent 13 years (2005-2018); pre-installation data were thus averaged for the preceding 13 years (1993-2005). Note there were not enough observations to calculate a standard deviation for pre-installation conditions at RU.

Name	**Pre-installation at the living shoreline, $g/cm^2/y$**	**Post-installation at the living shoreline, $g/cm^2/y$**	**Pre-installation at the reference shoreline, $g/cm^2/y$**	**Post-installation at the reference shoreline, $g/cm^2/y$**
Queens Landing (QL)	1.45±0.47	0.56±0.35 (p=0.20)	0.58±0.11	0.33±0.19 (p=0.14)
Oppenheim (OP)	0.94±0.25	1.30±0.53 (p=0.33)	1.49±0.80	0.49±0.11* (p=0.09)
Ruesch (RU)	0.84	1.08±0.60 NA	0.23±0.01	0.36±0.18 (p=0.22)
Hatton Garden (HG)	0.19±0.01	0.18±0.09 (p=0.18)	0.33±0.10	0.82±0.23* (p=0.01)
San Domingo (SD)	0.48±0.20	0.24±0.02* (p=0.01)	0.51±0.04	0.33±0.03* (p=0.008)
Environmental Concern (EC)	0.22±0.01	0.46±0.14* (p=0.003)	NA	NA
Myrtle Grove (MG)	0.37±0.01	0.66±0.37* (p=0.09)	0.34±0.18	0.62±0.11 (p=0.24)
Maritime Museum (MM)	0.50±0.02	0.54±0.13 (p=0.48)	0.44±0.04	0.50±0.04 (p=0.25)
Average	0.62±0.43	0.63±0.39 (p=0.98)	0.56±0.43	0.49±0.18 (p=0.73)

Table 4b. Pre- and post-installation mud content in the subtidal adjacent to living shorelines and corresponding reference shorelines. Data are averaged as described above.

Name	**Pre-installation at the living shoreline, %**	**Post-installation at the living shoreline, %**	**Pre-installation at the reference shoreline, %**	**Post-installation at the reference shoreline, %**
Queens Landing (QL)	25.2±12.4	66.0±7.4* (p=0.02)	49.7±0.6	49.4±13.6 (p=0.97)
Oppenheim (OP)	9.7±3.5	13.1±1.6 (p=0.38)	7.9±4.5	5.5±0.9 (p=0.31)
Ruesch (RU)	9.0	19.5±5.9 NA	30.3±2.8	22.6±1.2* (p=0.03)
Hatton Garden (HG)	45.2±4.1	41.5±2.5 (p=0.40)	90.9±3.8	91.5±1.1 (p=0.86)
San Domingo (SD)	88.7±3.9	90.1±0.9 (p=0.30)	75.4±2.2	79.6±4.3* (p=0.06)
Environmental Concern (EC)	89.6±5.2	92.5±1.7 (p=0.58)		

Myrtle Grove (MG)	3.2±0.8	17.2±19.0* (p=0.09)	3.9±3.2	28.2±16.8* (p=0.02)
Maritime Museum (MM)	2.4±0.1	3.7±1.1* (p=0.02)	3.9±0.2	37.0±34.5* (p=0.04)
Average	34.1±36.7	43.0±35.6 (p=0.11)	37.4±35.6	44.8±31.1 (p=0.25)

Table 4c. Pre- and post-installation organic content in the subtidal adjacent to living shorelines and corresponding reference shorelines. Data are averaged as described above.

Name	**Pre-installation at the living shoreline, %**	**Post-installation at the living shoreline, %**	**Pre-installation at the reference shoreline, %**	**Post-installation at the reference shoreline, %**
Queens Landing (QL)	1.2±0.6	3.3±1.7 (p=0.005)*	2.3±0.7	2.3±0.7 (p=0.88)
Oppenheim (OP)	2.3±1.3	2.3±1.5 (p=0.98)	0.5±0.2	0.6±0.1 (p=0.25)
Ruesch (RU)	2.0	1.6±0.8 NA	2.2±0.1	1.9±0.1 (p=0.22)
Hatton Garden (HG)	3.0±0.4	3.3±0.5 (p=0.40)	4.7±0.04	5.0±0.4* (p=0.07)
San Domingo (SD)	4.5±0.6	6.6±0.3 (p<0.001)*	7.5±0.4	8.3±1.2 (p=0.17)
Environmental Concern (EC)	4.3±0.3	4.0±0.1 (p=0.34)	NA	NA
Myrtle Grove (MG)	0.5±0.1	1.4±1.5 (p=0.15)	0.4±0.3	2.0±1.1* (p=0.02)
Maritime Museum (MM)	0.4±0.01	0.8±.03* (p=0.01)	0.4±0.3	2.7±3.7 (p=0.15)
Average	2.3±1.6	2.9±1.8 (p=0.11)	2.6±2.7	3.3±2.6 (p=0.10)