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Long-term performance and impacts of living shorelines in mesohaline Chesapeake Bay

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ABSTRACT

Shorelines in Chesapeake Bay, and many other estuaries and coastal embayments, are rapidly eroding, with even more rapid loss expected in the future from drivers like urbanization and accelerated relative sea-level rise (RSLR). Past efforts to stabilize shorelines using approaches like riprap or bulkheads generally have resulted in negative ecosystem impacts, resulting in a rise of ecosystem-based approaches using natural and nature-based features (NNBF) such as living shorelines, defined here as narrow marsh fringes with adjacent sills. Living shorelines provide similar ecosystem services as natural marshes but are threatened by the same stressors of environmental change, raising questions about their long-term resiliency and effectiveness in reducing shoreline erosion. Questions also remain about their potential impacts on benthic habitats in adjacent waters. These questions are especially relevant to the Chesapeake, where relatively rapid rates of RSLR and declining sediment supplies have led to widespread marsh loss, and submersed aquatic vegetation (SAV) is a critical indicator for water clarity. This study addresses these questions through field observations in ~10-year-old living shorelines and SAV habitats adjacent to them, along with observations at nearby reference (unaltered) shorelines. In general, shoreline erosion continued at or above historical rates at reference shorelines, but living shoreline installation builds shorelines seaward and results in net shoreline accretion. While the sand and organic content of bottom sediments in adjacent waters changed at many sites after living shoreline installation, changes were site-specific and typically in the same direction at both living and reference shorelines. These changes did not appear to impact SAV distributions, which followed regional trends likely linked to water clarity. Sediment and nutrient burial in the coastal zone, which includes both intertidal marsh and subtidal SAV habitats, was highest for living shorelines due to the addition of marsh habitat. While this study did not consider direct replacement of SAV with living shorelines, these results suggest that discouraging living shoreline installation in areas with SAV may miss an opportunity to enhance nutrient burial in the coastal zone.

1. Introduction

Coastal erosion is one of the most pressing environmental management issues around the world, with erosion rates expected to increase in the future from a variety of factors including accelerated rates of relative sea-level rise (RSLR), increased development, and increased storm intensity and/or frequency (Neumann et al., 2015; Williams et al., 2018). While shoreline erosion can be an important source of sediment needed to sustain some coastal habitats such as marshes and beaches, excessive sediment input can be detrimental to benthic organisms (e.g., light limitation and/or burial of oysters and submersed aquatic vegetation) (NRC, 2007; Noe et al., 2020). Erosion also threatens coastal infrastructure and private properties, resulting in efforts to stabilize shorelines. Historically, shorelines have been armored with structures such as seawalls and riprap (often referred to as “gray” approaches). Although these structures can have some positive ecosystem impacts (e.g., providing a hard substrate for oyster colonization), they generally result in negative impacts such as intensification of wave energy and fragmentation or loss of habitats (Bilkovic and Mitchell, 2013; Prosser et al., 2018). In response, ecosystem-based approaches (often termed “green”; e.g., restored or created marshes and oyster reefs) and hybrid approaches that combine engineering and ecological aspects (known as “natural and nature-based features; NNBF) to shoreline stabilization have risen in popularity and are even mandated in some states, including Maryland (Currin et al., 2010). However, much remains unknown about the long-term performance and impacts of these approaches.

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This paper focuses on living shorelines, which is a term that can be used to describe a wide range of NNBF. In this study, we define "living shorelines" as created fringing marshes with or without additional structures (Burke et al., 2005), which would be categorized as "engineered ecosystems" by (Chávez et al., 2021), specifically focusing on living shorelines with an adjacent rock sill. Living shorelines are generally installed with the goal of providing similar ecosystem services as natural marshes (e.g., reduced shoreline erosion and particulate input). This paper evaluates their effectiveness in reducing shoreline erosion and potential impacts on adjacent benthic habitats, especially for submersed aquatic vegetation (SAV). Collectively, SAV are keystone species in the Chesapeake Bay, providing effective nursery habitat for fish and other aquatic organisms (Beck et al., 2003), and their presence is a water quality indicator for restoration efforts (Batiuk et al., 2000). They are particularly responsive to light availability, and their abundance and distribution are limited by light reductions resulting from increased suspended sediment, as well as eutrophication-driven increases in phytoplankton and epiphytic algal growth (Kemp et al., 2004). Chesapeake Bay experienced catastrophic losses of SAV following Hurricane Agnes in 1972, after decades of slow decline (Kemp et al., 1983). The restoration of SAV in the Chesapeake Bay is part of the larger Bay restoration effort, leading to concern on the part of permitting agencies where proposed living shoreline construction may impact nearby current or potential SAV habitat (Nunez et al., 2022; Maryland Department of the Environment pers. comm).

Previous work has shown that shoreline armoring with bulkheads or rip rap is generally detrimental to SAV in the Bay (Patrick et al., 2014, 2016; Landry and Golden, 2018) but also can help improve sediment suitability for SAV in some places (Palinkas et al., 2017). Much less work has examined feedbacks between ecosystem-based approaches like living shorelines and SAV (while living shorelines were included in Palinkas et al. (2017), SAV was not present adjacent to any of the sites). These feedbacks are likely to mimic those between natural marshes and SAV. Marshes can benefit SAV by trapping sediment otherwise bound for adjacent waters and thus improve water clarity (Gurbisz and Kemp, 2014). However, marshes can also negatively impact SAV by becoming a source of sediment if they erode and/or drown with RSLR (Sanford and Gao, 2018). Likewise, SAV can help marshes by decreasing incoming physical energy and reducing marsh-edge erosion but could negatively impact marshes by trapping sediment needed for the marsh to maintain elevation (Carr et al., 2018; Nardin et al., 2018; Zhu et al., 2021). In addition, the inclusion of a structural element in many living shorelines introduces a boundary between marsh and SAV communities, the effects of which are not well understood. Vona et al. (2021) modeled sediment exchange between living shorelines and SAV, finding that sediment retention is highest when living shorelines with rock sills and SAV occur together. However, subtidal deposition in the absence of SAV was not quantified, and results were not compared to unaltered (reference) shorelines. Other recent work has shown that living shorelines can achieve functional equivalency to natural shorelines relatively quickly in many aspects (e.g., fish and infauna usage; Davenport et al., 2018, Guthrie et al., 2022) or even exceed benefits from natural marshes (e.g., storm erosion; Gittman et al., 2014). However, sediment and nutrient storage can take a decade or longer to reach equivalency (Chambers et al., 2021; Isdell et al., 2021), and integrating observations across the coastal zone (both subtidal and intertidal environments) is critical for understanding net storage (Huxham et al., 2018).

This study presents field observations at eight living shorelines, all of which contain a created marsh behind a continuous rock sill, half of which had adjacent SAV prior to installation. We examine living shoreline performance ~10 years after installation with respect to shoreline erosion and nutrient storage, and we assess potential impacts on SAV habitat and distributions in adjacent waters. Results from the subtidal zone are compared to those at nearby reference shorelines to help differentiate between changes related to environmental change versus living shoreline installation. We evaluate three main questions and related hypotheses. 1) Are living shorelines effective in reducing shoreline erosion? We expect that current shoreline erosion rates are equal to or greater than historical erosion rates at reference shorelines, but are greatly reduced at living shorelines. Sites with SAV should have lower historical shoreline erosion rates than those without SAV. 2) What are the impacts on SAV habitat and distributions in adjacent waters? We expect that living shoreline installation alters the benthic habitat in adjacent waters, but SAV is more sensitive to water quality than sediment type. After ~10 years, SAV distributions should not be affected by living shoreline installation. 3) Do living shorelines enhance sediment and nutrient storage in the coastal zone? We expect that net storage is highest at sites with both SAV and living shorelines. The answers to these questions are critical for management and regulatory agencies to help guide property owners. For example, current regulations in many places prohibit, or at least discourage, shoreline structures near or over SAV habitat, which can limit living shoreline implementation (Nunez et al., 2022). Chesapeake Bay Program (CBP) protocols for determining sediment and nutrient load reductions due to shoreline protection, important for meeting total maximum daily load requirements (TMDL) and/or potential future nutrient credits, first ask whether SAV is present and, if so, projects may not qualify as reductions because of a potential negative impact. The CBP workgroup also identified a critical gap in understanding structural impacts on SAV and a need to consider ecosystem changes rather than rely solely on sediment loads (Forand et al., 2017). While shoreline hardening is known to have negative impacts on Chesapeake SAV (Patrick et al., 2014, 2016; Landry and Golden, 2018), impacts from living shorelines are unknown.

2. Methods

2.1. Site selection and characterization

This study focused on mesohaline Chesapeake Bay, where shoreline erosion is the dominant sediment source to the nearshore (Hobbs et al., 1992), to minimize variability due to salinity, temperature, large storms, etc. To select sites, we produced a weighted overlay of SAV density data from aerial imagery by the Virginia Institute of Marine Science (VIMS) annual aerial SAV surveys (Orth et al., 2022) 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 projects between 2005 and 2008 provided by restoration managers and practitioners in the area and selected eight sites: four living shorelines with persistent SAV beds adjacent to the shoreline before installation and four living shorelines without SAV before installation (Fig. 1; Table 1). Each living shoreline was paired with a nearby (<0.5 km) unaltered shoreline as a reference site. All living shorelines were composed of a created marsh with a continuous rock sill at the toe (Fig. 2).

Historical shoreline-change rates were calculated from the Maryland Coastal Atlas (MCA; https://dnr.maryland.gov/ccs/coastalatlas/Pages/default.aspx), which contains data from shoreline surveys between 1841 and 1995 (Hennessee et al., 2002, 2003). The surveys most consistently available at our study sites were conducted in 1942 and 1994; however, data for one site (MG reference shoreline) were not available. Historical shoreline-change rates were calculated by importing the digitized shorelines from 1942 and 1994 into ArcGIS and measuring the difference at the shoreline perpendicular to the subtidal coring site (for consistency between living shorelines and reference shorelines), then dividing by the time elapsed. To calculate current shoreline-change 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 (first field survey year); shorelines were digitized and measured at identical points as above.

There were no studies, concurrently or previously, of hydrodynamics at these sites. We intentionally chose sites with similar broad-scale

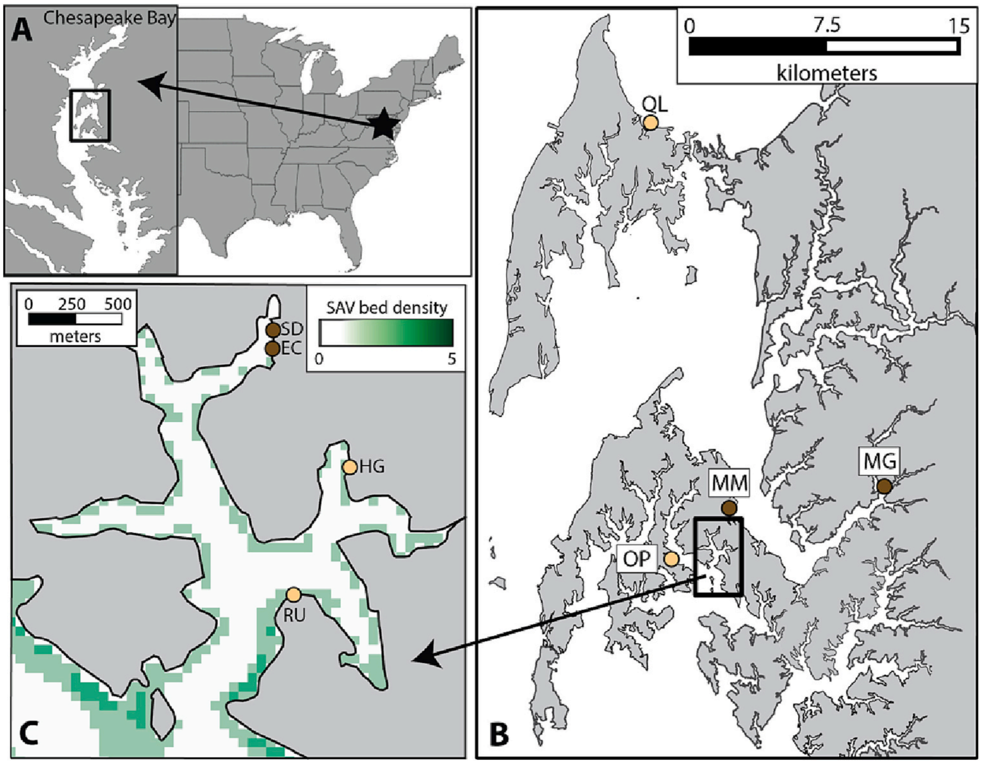

Fig. 1. (A) Map of states in the eastern USA; the star indicates the general region of Chesapeake Bay in the inset. The study area is outlined by the box and expanded in (B), in which light brown and dark brown circles indicate living shoreline sites with and without SAV before installation, respectively. C) Expanded view of some site locations and SAV weighted bed density from 1984 to 2017 (darker green = denser and more persistent SAV). Site abbreviations are listed in Table 1. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

Table 1
Name, install year, length, SAV status before and after installation (yes = persistent, dense; no = absent), REI (Relative Exposure Index), current (2003–2017) and historical (1942–1994) shoreline-change rate for living shorelines and corresponding reference sites. Note that negative rates indicate erosion; positive rates indicate accretion.

Name	Install Year	Length, ft. (m)	SAV before (after)	REI	Current shoreline-change, m/y	Historical shoreline-change, m/y
Queens Landing (QL)	2005	600 (182.9)	Yes (No)	278.83	+0.13	−0.04
Oppenheim (OP)	2006	440 (134.1)	Yes (Yes)	266.17	+0.68	−0.11
Ruesch (RU)	2008	1330 (405.4)	Yes (Yes)	241.20	+0.36	−0.225
Hatton Garden (HG)	2007	1860 (566.9)	Yes (No)	103.03	+0.79	+0.02
San Domingo (SD)	2007	770 (234.7)	No (No)	80.18	+0.74	+0.03
Environmental Concern (EC)	2005	550 (167.6)	No (No)	125.44	+0.52	−0.10
Myrtle Grove (MG)	2004	1500 (457.2)	No (No)	192.38	NA	−0.21
Maritime Museum (MM)	2008	615 (187.5)	No (No)	319.94	+0.02	−0.24
QL reference			Yes (No)	580.58	−0.40	−0.02
OP reference			Yes (Yes)	352.93	+0.14	−0.02
RU reference			Yes (Yes)	157.52	−0.08	−0.24
HG reference			Yes (No)	103.03	−0.11	−0.12
SD reference			No (No)	47.19	+0.50	0.0
EC reference			No (No)	48.69	−0.39	−0.12
MG reference			No (No)	181.75	NA	NA
MM reference			No (No)	798.13	NA	−0.45

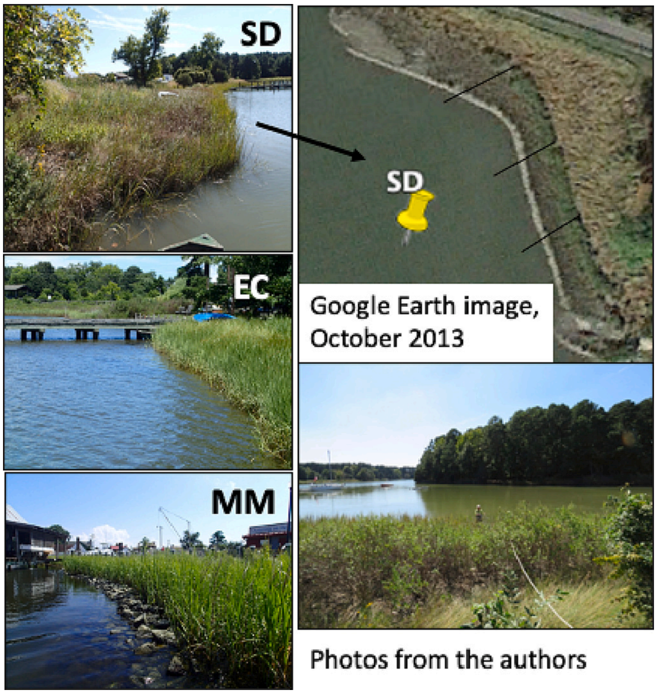

Fig. 2. (Left) Site photos of (top-bottom) SD, EC, and MM. (Left) Google Earth image of SD (top) showing transect lines. The 2013 image was chosen for image clarity. (bottom) Photo of transect marked by a measuring tape while sampling at SD.

weather patterns that drive the largest sources of energy and/or sediment supply. Tides in mesohaline Chesapeake Bay are microtidal (~0.3 m; Lee et al., 2017). However, wind-driven changes in water levels occur from changes on the Atlantic continental shelf (Chuang and Boicourt, 1989) and storms that can flood large areas for hours to days (Gong et al., 2009). Winds also create most waves in the mid-Bay, which are locally generated and fetch-limited (Lin et al., 2002). Sediment supply is mostly from shoreline erosion in this part of the Bay, but some sediment could be generated from overland flow with high rainfall (Hobbs et al., 1992). We quantified the fetch and relative exposure index as proxies of physical energy. These indices are often used as sole descriptors of hydrodynamic energy in site selection and design criteria (Hardaway et al., 2017). Fetch distance is often used as a proxy for physical energy, with a longer fetch implying higher wave exposure (Keddy, 1982). The wave climate of Chesapeake Bay is both fetch- and depth-limited (Hardaway, 1995), and so the fetch exposure and water depth (similar among study sites) provide an estimate of relative wave energy among sites. Fetch was quantified by measuring the distance to land along 60 vectors radiating from the subtidal core location at each site in Google Earth and then averaging all distances, including zeroes (Koch et al., 2006; Palinkas et al., 2016). This yields the average fetch and does not account for the dominant direction of the strongest winds. Thus, the average effective fetch and relative exposure index (REI) were also calculated for each site, following Fonseca et al. (2002) and Wong (2018). The effective fetch helps account for differences in shoreline morphology; the REI essentially weights the effective fetch by the percent of time the wind blew from a specific compass direction. We used the REI as the primary indicator of physical energy reaching individual sites. Hourly wind directions at Easton, MD (~15–20 km from all sites except QL; station ID 72404399999) were obtained from the National Centers for Environmental Information (NCEI; http://www.ncei.noaa.gov) from 2006 to 2017. For QL, hourly wind directions at the Bay Bridge Airport, MD (~5 km from QL; station ID 72038400124) were obtained for 2007–2017 (data were not available for 2006). Hourly wind observations from both data sets were averaged over the entire record to calculate REI. Note that these time periods represent post-installation conditions; corresponding wind data for pre-installation conditions were not available.

Historical SAV distributions were determined from the VIMS Interactive SAV Map (https://www.vims.edu/research/units/programs/sav/access/maps/index.php; Orth et al., 2022) from 1989 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 (categories assigned by VIMS as 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-min quadrangle ("quad"), in which the site was located.

2.2. Field surveys and associated laboratory analyses

Sediment and vegetation characteristics were assessed in the created marsh habitat of living shorelines and in the adjacent subtidal shallow-water SAV habitat. These surveys occurred in summer 2017 (EC, SD, HG, RU) and in summer 2018 (MG, MM, OP, and QL). At each living shoreline site, vegetation was assessed along three transects located perpendicular to the shoreline, 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 m 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 shorelines, only the subtidal zone was surveyed.

In the marsh, percent cover was estimated at each sampling point using a 0.25-m^2 quadrat marked off in a 10 cm × 10 cm grid (Bertness and Ellison, 1987). Marsh canopy height was estimated by measuring the five tallest stems at each point. All species at the point 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 sampling locations, one each in the high marsh, low marsh, and the transition zone between them. At EC and SD, all three stem density plots were located in the low marsh. 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 high marsh was characterized by *S. patens* and a variety of other species. In the subtidal, the presence or absence and species composition of SAV were recorded along each transect. Push cores (~20 cm long) were collected at the sampling points in the low marsh and high marsh, and vibracores (~3 m long) were collected in the subtidal adjacent to the living shoreline and the reference shoreline, along with push cores to capture relatively undisturbed surface sediments.

All cores were returned intact to the lab, where sediments were sectioned into 1-cm increments and then 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 h. Particulate nitrogen (N), phosphorous (P), and carbon (C) concentrations were determined for surficial sediments of push cores. 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) to 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 translated 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 were 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 and calculates discrete ages for each down-core depth horizon (Appleby and Oldfield, 1978). 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. This approach minimizes the inclusion of historical changes that may be present at the base of cores.

2.3. Statistical analyses

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 living shoreline and reference shorelines 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 subtidal sediment conditions. ANOVA was used to compare sediment characteristics among marshes and the two subtidal environments. We 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 sites and samples in this study was relatively low. Thus, for sediment analyses, we allowed a higher p-value of 0.10 to identify differences that may be physically meaningful. In assessing potential changes in subtidal sediment character after living shoreline installation, or the corresponding time horizon at reference shorelines, we highlighted trends with changes >20%, which is above the average replication error (Palinkas and Koch, 2012) and twice as high as the value (10%) used by Palinkas et al. (2022) in similar Chesapeake Bay environments.

We explored whether post-installation subtidal sediment conditions could be predicted with regression models. Parameters were chosen based on those most likely to be accessible to management and regulatory agencies before installation. These included REI, historical shoreline-change rate, shoreline type (living or reference), as well as pre-installation sand content, organic content, and sedimentation rates. Separate models were evaluated to predict post-installation sand content, organic content, and sedimentation rate. All parameters were included in the initial regression model and then removed stepwise to obtain the most parsimonious result.

3. Results

3.1. Site characteristics

At reference shorelines, historical shoreline-change rates ranged from −0.45 m/y to 0 m/y (no change), with an average of −0.14 ± 0.16 m/y (Fig. 2; Table 1). (Negative rates indicate erosion (shoreline moved landward) and positive change rates indicate accretion (shoreline moved seaward)). Current shoreline-change rates were similar to historical values (p = 0.83), ranging from −0.40 m/y to +0.50 m/y, with an average of −0.06 ± 0.46 m/y. At living shorelines, historical shoreline-change rates ranged from −0.25 m/y to +0.02 m/y, with an average of −0.11 ± 0.11 m/y. Current shoreline-change rates were all positive, ranging from +0.02 m/y to +0.78 m/y, with an average of +0.46 ± 0.30 m/y. Current rates were significantly higher ($p < 0.001$) than historical rates, but current and historical rates were not correlated (p = 0.13). Historical shoreline-change rates were not significantly different between living and reference shorelines, but current shoreline-change rates were much higher at living shorelines (p = 0.003). 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 (see Discussion).

The Relative Exposure Index (REI) was similar for living shorelines and reference shorelines (p = 0.29; Table 1). At reference shorelines, REI ranged from 47.19 to 798.13, averaging 281.69 ± 276.15. At living shorelines, REI ranged from 80.18 to 319.94, averaging 200.90 ± 89.49. Historical shoreline-change rates were negatively correlated with REI at living shorelines (R^2 = 0.32, p = 0.08) but not reference shorelines (p = 0.24). Current shoreline-change rates were also negatively correlated with REI at living shorelines (R^2 = 0.54, p = 0.03) but not reference shorelines (p = 0.49). Living shoreline length was not related to REI or historical or current shoreline-change rates. Note that negative shoreline-change rates in this paper indicate erosion, so a negative correlation between REI and shoreline-change rates means that erosion rates are higher when the relative exposure increases.

3.2. Vegetation observations

In the created marshes of living shorelines, the low marsh was dominated by *Spartina alterniflora* (cordgrass), but *Phragmites australis* (phragmites) was dominant along 2 of the survey transects at QL (Table 2). The high marsh was typically more diverse, with *S. patens* (saltmarsh hay) dominant at a majority of sites, but other species were co-dominant at some sites. Mean percent cover in the marshes was fairly uniform, ranging ~75–100% at all sites, with little difference between high and low marsh or among sites. Stem height was similarly uniform,

Table 2
Summary of vegetation surveys in summer 2017 and summer 2018. The first two data columns list the stem density and dominant species in the low marsh (first line) and high marsh (second line) in living shorelines. The last two columns list the dominant SAV species and percent vegetated sites per transect in the subtidal adjacent to living shorelines (LS) and reference shorelines (ref). NP = SAV was not present. Species are *S. alterniflora* (*S. alt.*), *S. patens* (*S. pat.*), *Bulboschoenus robustus* (*B. rob.*), *P. australis* (*Phrag.*), *I. frutescens* (*Iva*), and *R. maritima* (*Ruppia*).

Site	Marsh stem density (m^{-2})	Marsh dominant species	Dominant SAV species – LS (%vegetated)	Dominant SAV species – ref. (%vegetated)
QL	190.0	*Phrag., S. alt.*	NP	NP
	469.4	*Phrag.*	0	0
OP	246.7	*S. alt.*	*Ruppia*	*Ruppia*
	1652.5	*S. pat., Iva*	89.7 ± 5.2	73.8 ± 5.1
RU	930.0	*S. alt.*	*Ruppia*	*Ruppia*
	2158.3	*S. pat.*	85.7 ± 0	47.6 ± 12.6
HG	610.0	*Phrag., S. alt.*	*Ruppia*	*Ruppia*
	1341.7	*S. pat.*	52.4 ± 12.6	76.2 ± 9.6
SD	255.6	*S. alt.*	*Ruppia*	*Ruppia*
	NA	*S. pat.*	55.6 ± 11.1	27.8 ± 14.7
EC	412.2	*S. alt.*	*Ruppia*	*Ruppia*
	NA	*S. pat., B. rob.*	35.6 ± 2.2	33.3 ± 0
MG	330.0	*S. alt.*	NP	NP
	3706.7	*S. pat.*	0	0
MM	330.0	*S. alt.*	*Ruppia*	NP
	2272.5	*S. pat.*	6.7 ± 6.7	0

ranging 125–175 cm. Although the stems of some species found in the high marsh were decumbent or prostrate (e.g., *S. patens*), their lengths were similar to the upright stems of *S. alterniflora* in the low marsh. In addition, taller species such as phragmites and *Iva frutescens* (high tide bush) were sometimes found in the high marsh, 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 (Table 2). Lower stem densities were associated with *S. alterniflora* in the low marsh, while higher densities were associated with *S. patens* and *Distichlis spicata,* which have finer stems than *S. alterniflora*, in the transition and high marsh zones.

In the subtidal, SAV was adjacent to 6 of the 8 living shorelines and 5 of the 8 reference shorelines (Table 2; Fig. S1). Note that these data reflect SAV observed at the time of the survey, which may differ from those in datasets with more years and/or from aerial surveys (e.g., Table 1). *Ruppia maritima* (widgeon grass) was the dominant subtidal species at all sites, occurring exclusively at most sites, along with *Zannichellia palustris* (horned pondweed) at EC, HG and HGref. No SAV was found at QL, QLref, MG, MGref, or MMref, and just one occurrence of *Ruppia* was found at MM. While there was variability in the percent vegetated sites between the pairs of living shorelines and reference shorelines, there were no significant differences ($p = 0.05$) among the pairs.

SAV distributions in the region varied considerably over time, exemplified by the SAV area in the quad containing most of our sites (Fig. 3). Here, SAV was low to non-existent at the beginning of the record, increased to a peak in 1997, subsequently declined to a period of low to no SAV in the mid-late 2000s, and then rebounded to a secondary peak in 2017. SAV at most sites followed trends in their respective quad, both before and after installation (e.g., RU; Fig. 3, top), with three exceptions. First, at HG, which was in the same quad as RU, SAV disappeared from the site and the quad in 1999, but SAV did not rebound at the site (as of 2019) as it did in the quad (Fig. 3, bottom). Note that while we observed SAV at HG in 2018, the area was not completely mapped via aerial photography that year. SAV at HGref also followed this pattern, except for 2015 and 2016. Second, SAV disappeared at QL and QLref after the installation year, although SAV was present in 2019 at both sites. Lastly, MG and MGref, and MM and Mref, did not have SAV at any point in the record, even though the quad had occasional SAV. Direct comparisons are challenged by the differing nature of data for the quad (discrete area) and site (bed density classes). Even so, at individual sites, SAV density class was correlated with SAV area in the corresponding quad, aside from the noted exceptions, whether considering all years together or separating observations into years before and after installation (Table S1). Sites with persistent SAV before installation had more robust statistics, since they had more non-zero observations. SAV density class was similar at sites adjacent to living shorelines and paired reference shorelines, for all years together and when years before and after installation were considered separately.

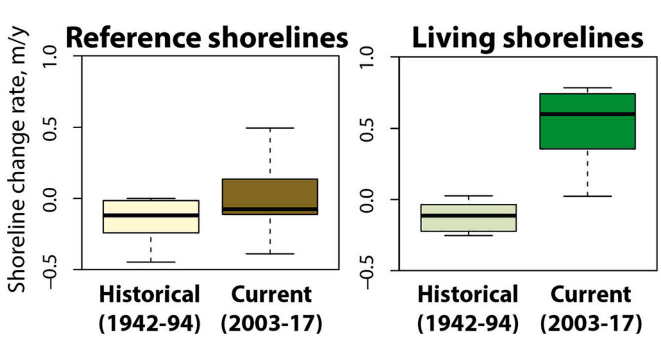

Fig. 3. Shoreline-change rates at reference shorelines (left) and living shorelines (right). In each, historical rates are on the left and current rates are on the right. Each box shows the range of values, and the solid line and error bars represent the median and standard deviation, respectively. Negative rates = erosion; positive rates = accretion.

3.3. Surface sediment characteristics and seasonal-scale sedimentation rates

Characteristics of surface sediments (topmost 1 cm of cores) within each environment at individual sites are listed in Table S2 and averaged in Table 3. Sand content varied the most (highest standard deviations), ranging from 37.3 to 97.8% in marshes, 18.4–92.6% in the subtidal adjacent to living shorelines, and 4.1–88.8% in the subtidal adjacent to reference shorelines. Note that "sand" refers to material retained on a 64-μm sieve, whether it is organic or inorganic. Marshes had similar sand content as the subtidal zones adjacent to them ($p = 0.62$). The subtidal zone adjacent to living shorelines had higher sand content than the subtidal adjacent to reference shorelines ($p = 0.02$). Some individual sites had quite different sand content in different environments; e.g., for RU, sand content was 37.7% in the marsh, 80.1% in the subtidal adjacent to the living shoreline, and 60.9% in the subtidal adjacent to the reference shoreline. Organic content was much higher in the marshes (average 14.6 ± 8.6%) than in the subtidal adjacent to living shorelines (average 4.9 ± 3.2%; $p = 0.01$). Subtidal sediments adjacent to living shorelines had higher organic content than those adjacent to reference shorelines (p = 0.02). PC and PN were also higher in the marshes than in the subtidal adjacent to living shorelines ($p = 0.04$ for both); subtidal PC and PN were similar adjacent to living shorelines and reference shorelines ($p = 0.27$ for PC, $p = 0.28$ for PN). PP was similar among the three environments ($p > 0.13$ for all).

Seasonal (^{7}Be-derived) mass deposition rates also varied widely within each environment, resulting in similar average values ($p > 0.60$ for all) among the three environments. ^{7}Be was not detected at two sites in the subtidal adjacent to living shorelines (HG and EC) and one site in the subtidal adjacent to reference shorelines (RU), which could reflect erosion or negligible deposition. The mass deposition rate at QL was ~2–3 times higher than the average rate plus one standard deviation in all environments. If QL were excluded, average mass deposition rates would be 0.51 ± 0.20 g/cm^2/y in the marshes, 0.28 ± 0.37 g/cm^2/y in the subtidal adjacent to living shorelines, and 0.44 ± 0.40 g/cm^2/y in the subtidal adjacent to reference shorelines. Removing QL from the full data set did not affect statistical relationships among average deposition rates ($p > 0.25$ for all) or trends in sediment character described above. Mass deposition rates are translated to linear rates in Table 3 via the bulk density to facilitate comparison with RSLR in the Discussion.

Nutrient burial rates (product of nutrient content and mass sediment deposition rate) are summarized in Table 4. QL had relatively high nutrient concentrations in both living shoreline and reference subtidal environments (nutrients were not measured in the marsh at QL) that, along with high sedimentation rates, resulted in very high nutrient burial rates that biased average values. Including QL, there were no significant differences in PC, PN, or PP burial rates among the three environments. Excluding QL, PC and PN rates in the marshes were significantly higher than in the subtidal adjacent to living shorelines ($p = 0.07$ and $p = 0.08$, respectively), but the two subtidal zones have similar PC and PN rates. There were no differences in PP among the three environments.

3.4. Changes in subtidal sediments associated with living shoreline installation

Changes in subtidal sediments were observed in at least one parameter at most individual sites, whether indicated by statistics or changes above 20% (Fig. 4). Sand content decreased at 2 living shorelines and 3 reference shorelines, with no change occurring at the other 6 living shorelines and 3 of the reference shorelines. Sand content increased at 1 reference shoreline (RUref). Organic content increased at 4 living shorelines and 4 reference shorelines, with no change occurring at 3 living shorelines and 3 reference shorelines, and a decrease

Table 3
Average characteristics of surface sediments and seasonal sediment deposition rates in the three environments. For subtidal nutrients, data at the top of cells include all sites, and data at the bottom exclude QL. For deposition rates, mass rates are listed at the top of the cell, and corresponding linear rates are listed in parentheses at the bottom. Subtidal deposition rates do not include OP and MM. PC = particulate carbon, PN = particulate nitrogen, PP = particulate phosphorus.

	Sand%	Organic%	PC%	PN%	PP%	Deposition rate, g/cm^2/y (cm/y)
Living shoreline marshes	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)
Subtidal – living shorelines	65.8 ±29.9	4.9 ±3.2	1.78 ±2.96 (0.76 ±0.75)	0.17 ±0.26 (0.08 ± 0.08)	0.43 ±0.26 (0.43 ±0.28)	0.42 ± 0.49 (0.87 ± 1.05)
Subtidal – reference shorelines	45.5 ±31.0	2.7 ±2.5	2.41 ±2.99 (1.48 ±1.53)	0.24 ±0.27 (0.15 ±0.15)	0.66 ±0.56 (0.67 ±0.60)	0.86 ± 1.07 (1.31 ± 1.37)

Table 4
Average nutrient burial rates in the three environments. Data at the top of cells include all sites except OP and MM; data at the bottom also exclude QL.

	PC burial, mg/cm^2/y	PN burial, mg/cm^2/y	PP burial, mg/cm^2/y
Living shoreline marshes	32.0 ± 28.3	2.8 ± 2.5	9.3 ± 11.4
Subtidal – living shorelines	18.3 ± 41.6 (1.3 ± 1.3)	1.7 ± 3.7 (0.16 ± 0.15)	1.6 ± 2.2 (0.74 ± 0.82)
Subtidal – reference shorelines	54.5 ± 102 (13.4 ± 18.8)	5.1 ± 9.4 (1.3 ± 1.9)	6.3 ± 5.9 (4.2 ± 0.74)

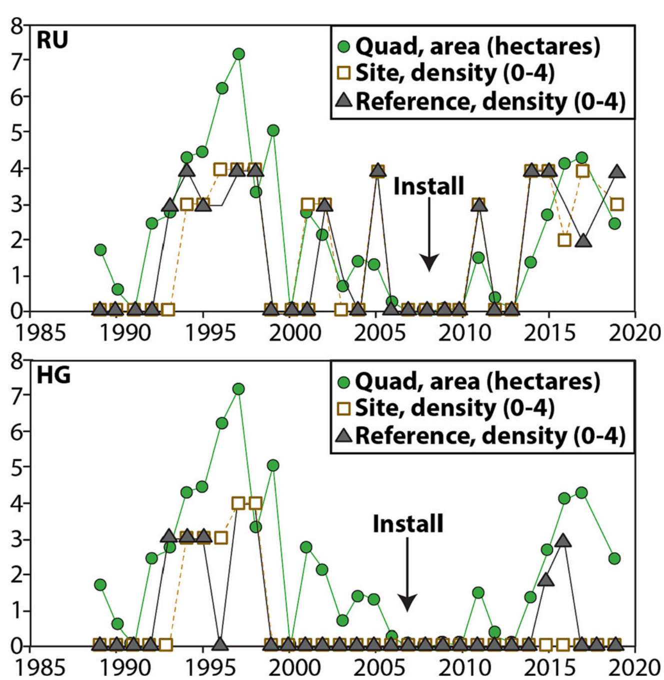

Fig. 4. Historical SAV data for RU (top) and HG (bottom). Green circles show SAV area in the quad, which is the same for both. Open brown squares and gray triangles show SAV bed density classes at the living shoreline and reference shoreline, respectively. The arrow points to the year of the living shoreline installation. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

occurring at 1 living shoreline (RUref). The difference in organic content was always in the opposite direction as sand content; however, there were only two sites with changes in both parameters (QL and MGref). Mass accumulation rates increased at 4 living shorelines and 3 reference shorelines, decreased at 2 living shorelines and 3 reference shorelines, and did not change at 2 living shorelines and 1 reference shoreline. There was no clear relationship between changes in sediment character and accumulation rates. For example, at QL, sand content decreased (organic content increased), but the accumulation rate decreased. At MG, sand also decreased (no change in organic) but the accumulation rate increased. Only QL and MGref had meaningful changes in all three parameters. If a meaningful change occurred, it was usually in the same direction for the living shoreline and its paired reference shoreline. The exception was that the accumulation rate increased at OP but decreased at OPref. (Fig. 5.)

Overall, subtidal sediment characteristics did not differ between living shorelines and their paired reference shorelines before or after installation (Tables 5, S3). All parameters had similar ranges whether they were grouped by shoreline type (living or reference) or age (pre- or post-installation). Sand content varied from ~10–95%, organic content varied from ~0.5–7%, and mass accumulation rates varied from ~0.25–1.0 g/cm^2/y. As a result, there were no significant differences in pre- versus post-installation sand content ($p = 0.11$), organic content ($p = 0.11$), or mass accumulation rates ($p = 0.98$) at living shorelines or reference shorelines ($p = 0.25$, $p = 0.11$, $p = 0.73$, respectively).

3.5. Relationships among variables

Sites with SAV had similar REI, historical shoreline-change rates, and current shoreline-change rates as those without SAV, whether at living shorelines or reference shorelines. There were also no significant differences in subtidal sediment characteristics or sedimentation rates when sites were grouped as having SAV present versus absent before or after installation for living or reference shorelines. Thus, for the following statistical analyses, a more robust parameter was used to quantify vegetation: the percent of vegetated sites (direct observations) for comparison with push-core data (Table S2), and the percent of years with vegetation present in aerial photographs (using the same years as represented by pre- and post-install sediment data) for comparison with long-core data (Table S3).

For push-core data, at living shorelines, REI was negatively correlated with the current shoreline-change rate ($p = 0.04$) and marsh sediment organic content ($p = 0.003$) (see Fig. S2 for correlation matrices). Marsh sand content was negatively correlated with organic content ($p = 0.009$) and all 3 nutrient parameters ($p = 0.06$, $p = 0.07$, and $p = 0.09$ for PP, PN, and PC, respectively). In the subtidal adjacent to living shorelines, REI was positively correlated with sand content ($p = 0.01$) and sedimentation rate ($p = 0.03$), and sand content was negatively correlated with PP ($p = 0.008$). In the subtidal adjacent to reference shorelines, REI was also positively correlated with sand content ($p = 0.02$) and sedimentation rates ($p = 0.008$), and sand content was negatively correlated with PP ($p < 0.001$). REI was negatively correlated with PP ($p = 0.07$). Sedimentation rates were positively correlated with PC and PN ($p < 0.001$ for both) and negatively correlated with the percent vegetated sites ($p = 0.09$). However, all environments at QL had

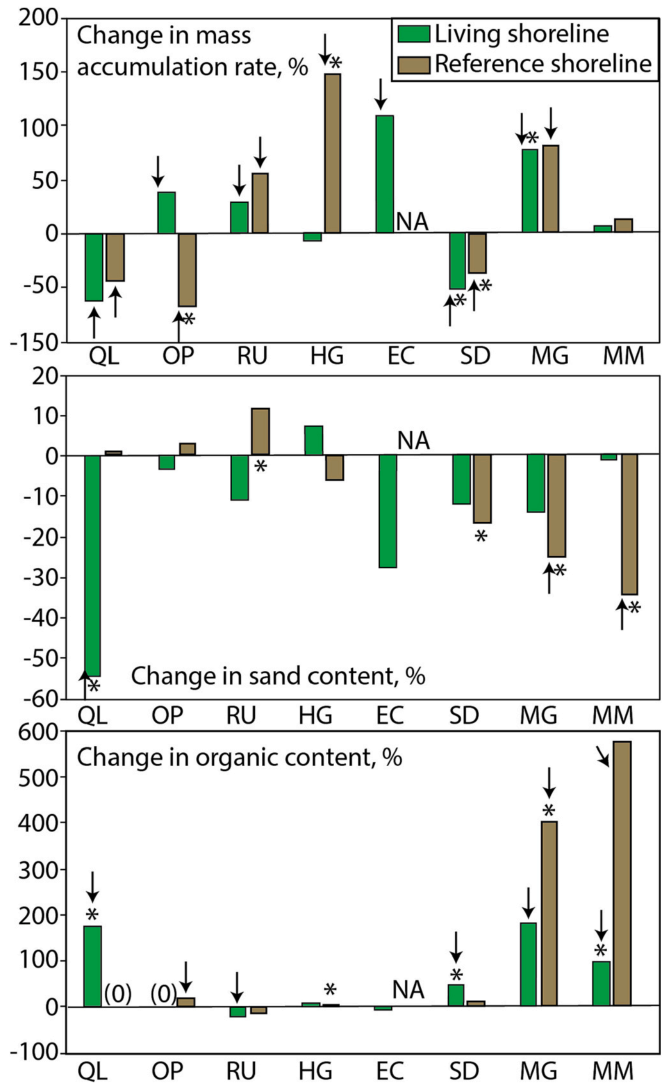

Fig. 5. Changes in average sediment (top to bottom) accumulation rate, sand content, and organic content observed at subtidal sites. Sites are listed as in Table 1. Asterisks (*) indicate $p < 0.10$ in t-tests, arrows indicate changes >20%.

Table 5
Subtidal sediment characteristics averaged over pre- and post-installation time periods (see text for details). For accumulation rates, mass rates are listed at the top of cells, and the corresponding linear rates are listed at the bottom. Statistical comparisons were made only for pre- versus post-installation conditions and not between data at living versus reference shorelines. The p-value for accumulation rates is for mass rates only.

	Pre-installation at the living shoreline	Post-installation at the living shoreline	Pre-installation at the reference shoreline	Post-installation at the reference shoreline
Sand content, %	65.9 ± 36.7	57.0 ± 35.6 p = 0.11	62.6 ± 35.6	55.2 ± 31.1 p = 0.25
Organic content, %	2.3 ± 1.6	2.9 ± 1.8 p = 0.11	2.6 ± 2.7	3.3 ± 2.6 p = 0.10
Accumulation rate, g/cm^2/y (cm/y)	0.62 ± 0.43 *(0.45 ± 0.28)*	0.63 ± 0.39 *(0.54 ± 0.24)* p = 0.98	0.56 ± 0.43 *(0.46 ± 0.26)*	0.49 ± 0.18 *(0.64 ± 0.26)* p = 0.73

anomalously high sedimentation rates and PC, PN, and PP. Removing QL from the marsh dataset only caused one change in correlations, with stem densities becoming positively correlated with sedimentation rates (p = 0.008). In the subtidal adjacent to living shorelines, without QL, sedimentation rates were not correlated with REI (p = 0.76) or current shoreline-change rates (p = 0.27). However, PC and PN were negatively correlated with REI (p = 0.07 and p = 0.09, respectively) and sand content (p = 0.007 and p = 0.02, respectively), and all the nutrient parameters were positively correlated ($p < 0.04$ for all). The only difference in the subtidal adjacent to reference shorelines (without QL) was that sedimentation rates were not correlated with REI (p = 0.76). The length of living shorelines was not correlated to any other parameter, with or without QL.

For long-core data, REI was negatively correlated with the current (p = 0.04, as above) and historical (p = 0.08) rates of shoreline change for living shorelines but not reference shorelines. Before installation, in the subtidal adjacent to living shorelines, organic content was correlated with the shoreline-change rate (positive), REI (negative), and sand content (negative). In turn, sand content was also correlated with shoreline-change rate (negative) and REI (positive). The sedimentation rate was positively correlated with REI. The average SAV bed density was not correlated with any of these parameters. In the subtidal adjacent to reference shorelines, sand and organic content were negatively correlated, and sedimentation rates and average SAV bed densities were positively correlated. After the installation period, adjacent to living shorelines, organic content was correlated with the pre-install organic content, along with its correlated parameters (historical shoreline-change rates, REI, pre-install sand content), and post-install sand content. Similarly, post-install sand content was correlated with the pre-install sand content and its correlated parameters. However, the post-install sedimentation rate was correlated with only the post-install average SAV bed density. After the installation period, adjacent to reference shorelines, organic content was correlated with the pre-install organic and sand content, and the post-install sand content. Sand content was correlated with the pre-install sand and organic content, and the post-install organic content. Sedimentation rates were not correlated with any single parameter. Average SAV bed densities before and after installation were correlated. Lengths of living shorelines were not correlated with any parameter.

All sites were combined to assess multiple linear regression models that included shoreline type (living versus reference) as a parameter, focusing on predicting post-installation sediment characteristics from information that should be readily accessible to managers. The post-install sedimentation rate could be predicted by combining the historical shoreline-change rate and average pre-installation SAV bed density (p = 0.04, R^2 = 0.35). The post-install organic content could be predicted solely by the pre-install organic content ($p < 0.001$, R^2 = 0.80) or sand content (p < 0.001, R^2 = 0.63), but if pre-existing sediment data were not available, it could also be predicted by the historical shoreline-change rate and average pre-install SAV bed density (p = 0.08, R^2 = 0.24). Similarly, the post-install sand content could be predicted solely by the pre-install sand (p < 0.001, R^2 = 0.83) or organic (p < 0.001, R^2 = 0.55) content, as well as by combining the historical erosion rate and the average pre-install SAV bed density (p = 0.05, R^2 = 0.31). Note that shoreline type (living or reference) was not a significant parameter in any regression model.

4. Discussion

The results presented above are synthesized to evaluate the three main research questions and hypotheses in the sections below.

4.1. Shoreline change

The first question asked whether living shorelines reduce shoreline erosion. This is perhaps the most important question since erosion reduction is one of the main factors influencing the choice of living shorelines for shoreline stabilization (Stafford and Guthrie, 2020). To address this, we calculated historical and current rates of change in shoreline position at locations of living shorelines and compared them to observations at paired reference shorelines that did not have any shoreline-stabilization structures. The relative exposure index (REI) was similar between living shorelines and references, indicating that the physical dynamics driving shoreline change were similar at both shoreline types, and this is reflected in the similarity of historical shoreline-change rates among sites. We found that historical shoreline-change rates were mostly negative, indicating erosion, with comparable values between living and reference shorelines. However, after living shoreline installation, most reference shorelines continued to erode at or above historical rates, while most living shorelines showed lateral movement (seaward shoreline change).

The continued erosion at reference shorelines suggests that the energy regime did not have large-scale changes, and so the accretion at living shorelines is more likely linked to installation, during which the land-water interface is moved seaward. While the REI does not include many factors that influence the energy reaching the shoreline (e.g., boat wakes; Bilkovic et al., 2019), most factors should have a similar effect on both shoreline types given the close proximity of sites. Living shorelines are often constructed with fringing marshes and a structural component (e.g. rock sills) along the toe of the marsh to attenuate incoming wave energy (Smith et al., 2020; Guthrie et al., 2022). In many cases, including the sites considered here, the shoreline is graded from elevations supporting existing upland vegetation to those needed for high- and low-marsh plants by building the shoreline seaward, placing clean sand fill to attain appropriate elevations within the tidal frame, and then creating the marsh via planting (Bilkovic and Mitchell, 2017; Polk and Eulie, 2018). Assessing shoreline positions before and after installation in these cases is relatively straightforward since the rock sill is essentially fixed. However, if the created marsh is unable to keep up with relative sea-level rise (RSLR) and/or experiences erosion, a gap can develop between the sill and the vegetation as has been observed at a 14-year-old living shoreline in the Maryland Coastal Bays (Sun et al. 2022). While we did not measure the distance between the sill and marsh vegetation at the living shorelines in this study, we did not observe obvious gaps.

The average rate of RSLR between 1971 and 2002 measured in nearby Cambridge, MD (station 8,571,892; http://tidesandcurrents.noaa.gov) was 3.0 mm/y, increasing to an average rate of 5.0 mm/y from 2003 to 2017. This time span is less than the 19-y period of the metonic cycle and could be biased. Extending the time span to the 19 years before sampling (1998–2017) yields an average rate of 4.7 mm/y. Three of our sites (OP, MM, MG) had linear marsh deposition rates (units of cm/y; shown in Tables 3, 5, and S2) below the 2003–2017 rate of RSLR, with the rates at OP and MM being within <1 mm/y of RSLR. The deposition rate at MG suggests that it has been trapping about half as much sediment as it needs to be sustainable, although there was no discernable gap between the sill and marsh vegetation at this or any other site. It is important to keep in mind that ^{7}Be measurements reflect a much shorter time scale (seasonal) than RSLR (decadal), and caution should be used in directly comparing rates. Cores were collected during the growing season when plants can enhance sedimentation via trapping and/or organic matter production. Annual-scale deposition rates integrate periods of non-deposition/erosion in the winter, when plants are absent and/or physical conditions are more energetic. If the seasonal deposition rates reported here are considered as maximum or perhaps over-estimated rates, then most of these marshes are not keeping up with RSLR and may not be sustainable over longer time scales. Elevation changes were not measured in marshes but would be useful to help resolve these discrepancies and better predict the future trajectory of the marshes.

It is difficult to place our shoreline observations into a broader regional context, given that the most recent shoreline data for the counties in which these sites are located (Talbot, Queen Anne's) are from the 1990s, and the most recent inventory of shoreline structures was completed in 2004 (Berman et al., 2006). Palinkas et al. (2022) quantified shoreline changes over similar time periods in two small creeks in Talbot County, finding that average shoreline-erosion rates decreased from 1942 to 1994 to 2003–2015, but the degree and timing of shoreline stabilization after 2004 were unknown. In the broader region of upper Chesapeake Bay, shoreline erosion also decreased over time, but insights were hindered by the same lack of recent data (Russ and Palinkas, 2020). Shoreline positions over broad spatial and temporal scales can be discerned from historical data and aerial imagery (Stockdon et al., 2002; Eulie et al., 2018; Johnson et al., 2020; Polk et al., 2021). However, it is difficult to determine whether changes in those positions arise from reduced shoreline-erosion rates or the installation of shoreline-stabilization structures without an updated shoreline inventory, direct observations, or other site-specific knowledge. More current data are available for Talbot County (Nunez et al., 2021) that provide important updates on current shoreline practices and general characterizations but not the timing of stabilization or discrete shoreline-change rates. Living shorelines reduced shoreline-erosion rates at several living shorelines in North Carolina (Polk et al., 2021), though studies were hindered by similar limitations as above.

Shoreline-change rates were unrelated to average SAV bed density before or after the period of living shoreline installation at either shoreline type. This is somewhat surprising given feedbacks between SAV and sediment. SAV attenuates wave energy reaching the shoreline (Koch, 2001; de Boer, 2007; Vona et al., 2021), so shorelines with sparse SAV should have higher erosion rates in similar energy regimes. Conversely, since shoreline erosion is the dominant source of sediments in mesohaline Chesapeake Bay (Hobbs et al., 1992) and excess sediment can be detrimental to SAV (Koch, 2001; Kemp et al., 2004), shorelines with high erosion rates should have sparse SAV. Neither of these relationships was evident in our results, although insights are limited by the relatively small number of sites and the lack of SAV data before 1979. In addition, site selection was biased by the tendency of management and regulatory agencies to favor installation of living shorelines in relatively low-energy environments and in areas without robust SAV (Hardaway et al., 2017). Whether feedbacks between shoreline erosion, sediment, and SAV are more robust in areas with higher energy and/or where SAV is more abundant and denser is an important area for future research.

4.2. SAV habitat and distributions

The second question focused on the potential impacts of living shoreline installation on SAV habitats and distributions in adjacent waters. Living shorelines effectively reduce shoreline erosion and thus sediment input to adjacent waters. Since most shorelines in the mesohaline Bay are sandy (Halka, 2005), adjacent subtidal sediments could become finer and more organic, with reduced sedimentation rates after installation. In contrast, erosion of reference shorelines was similar over time, suggesting that adjacent subtidal sediment characteristics and accumulation rates should be similar over time.

In this study, all sites, whether they were adjacent to living shorelines or references, showed changes in at least one sediment parameter, except the living shoreline at HG. MG was the only location with changes in all 3 parameters at both living and reference shorelines. The magnitude and direction of changes varied among sites, instead of any systematic changes related to installation timing or shoreline type, but changes were usually in the same direction or not at all between living shorelines and their paired reference shorelines. The only exception was at OP, where the accumulation rate increased at the living shoreline but decreased at the reference shoreline, with no changes in sand or organic content at either site. Interestingly, the shoreline-change rate at the reference shoreline shifted from slightly erosional (−0.02 m/y) before the installation period to accretional (+0.14 m/y) afterward. The reference shoreline is located downstream of the living shoreline, and a creek enters the water between them. The mouth of the creek can be seen widening in aerial imagery, and it is possible that the locus of sedimentation shifted landward over time, decreasing sediment input to the subtidal coring site at the reference shoreline.

Since changes in sediment characteristics were in the same direction at the living and reference shorelines, changes in sediments were likely driven by larger-scale regional processes and did not result from living shoreline construction. For example, most sites had decreasing sand and increasing organic content, which is consistent with the general trend of increasing fine and organic sediments in the Chesapeake Bay (Kemp et al., 2005) and observations in other regional SAV beds (Palinkas and Koch, 2012). Whether this trend is driven by increasing inputs of fine suspended sediment or decreasing inputs of coarser material is an area ripe for future research. While modeled suspended-sediment loads have been decreasing in the Choptank River, unlike other non-tidal Chesapeake tributaries (Zhang and Blomquist, 2018), the discharge is measured well upstream of our study sites and little is known about trends in the small creeks surrounding the main stem.

The presence of SAV was not related to sediment characteristics (sand and organic content), in contrast to previous work suggesting that SAV prefers sediments that have $<5\%$ organic content and $>$ 28.0–99.6% sand (Koch, 2001). Mean values of sand and organic content were within these ranges, pre- and post-installation, for all sites except the SD and EC living and reference shorelines and the HG reference shoreline (Table 4). While the lack of SAV in aerial imagery of the EC and SD sites may agree with SAV substrate limits, SAV was observed during ground surveys at these sites, and the HG reference shoreline had robust SAV prior to the installation period when mean sand content was ~9%. SAV disappeared after the installation period, and organic content increased near the limit for SAV (with a mean value of 5.0%), but sand content remained the same. SAV was more prevalent in ground surveys (11 sites) than aerial imagery (8 sites pre-install, 4 sites post-install), likely due to different sampling times and resolutions (Orth et al., 2022). For example, subtidal sediment conditions seem suitable adjacent to the MG and MM living and reference shorelines, but SAV has never been observed in aerial imagery of these sites nor was SAV observed in our ground surveys, except for the survey at MM living shoreline. SAV has been present in the corresponding quads, suggesting that SAV could be limited by seed transport and/or poor water quality rather than sediment characteristics (Kemp et al., 2005; Orth et al., 2017). Neither sand content nor organic content was correlated with average SAV bed density before or after installation, regardless of shoreline type, suggesting that the SAV species observed (primarily *Ruppia maritima*) were not sensitive to sediment characteristics. Instead, SAV bed densities at individual sites were simply correlated with SAV in the corresponding quad, both before and after installation. This dependence on regional water quality rather than modifications on adjacent land was also observed in lower Chesapeake Bay (Blake et al., 2014). However, many more study sites are required to adequately characterize the relationship of SAV to ecosystem-based approaches like living shorelines.

Accumulation rates increased post-installation at most sites (excluding the QL living shoreline and the SD living and reference shorelines). This could be a response to accelerating RSLR, reflecting increasing accommodation space, which increases the inundation depth and duration and thus the opportunity for sedimentation, producing higher accumulation rates. It could also reflect potential biases in the model used to calculate sediment ages and corresponding accumulation rates, especially in deeper sections of cores where sediment ages are more uncertain (Sanchez-Cabeza et al., 2000; Appleby, 2008; MacKenzie et al., 2011). Deeper sections may also have underestimated sedimentation rates since sedimentation rates often decrease as the time interval over which they are averaged increases (Sadler, 1981). These biases are minimized by using equivalent time periods to compare pre- and post-installation conditions at each site. A related study using the same approach found that sedimentation rates actually decreased after the installation of shoreline-stabilization structures (Palinkas et al., 2017), indicating that the direction of change is not related to methodological bias. Accumulation rates were generally positively related to average SAV bed densities, although the correlation was only significant before installation at reference shorelines and after installation at living shorelines. This reflects enhanced trapping of sediment by SAV; the relationship would likely be more robust in areas with denser and more persistent SAV.

The site-specific nature of these results challenges efforts to anticipate changes in the adjacent subtidal habitat after installation. To help predict changes after installation, we combined all sites into regression models and used data that should be accessible to practitioners and/or management and regulatory agencies. Shoreline type (living or reference) was included in these models but was not a significant parameter in any of them. Not surprisingly, most variability in post-installation sand and organic content was explained by the pre-installation sand and organic content, in any combination. And, the best predictor of post-installation SAV was pre-installation SAV. This further supports our conclusion that changes in these parameters are driven by larger-scale processes and not living shoreline installation. However, pre-installation sediment data may not available, in which case post-installation sand content, organic content, and sedimentation rates could all be predicted by combining the historical shoreline-change rate and the average pre-installation SAV bed density. While the regressions only explained 25–35% of the variability, they provide a tractable way to anticipate potential changes. Local geomorphic processes, such as the widening creek mouth between the living shoreline at OP and the corresponding paired reference shoreline, likely explain more of the variability but require complex models (e.g., Vona et al., 2021) not readily accessible to practitioners or management and regulatory agencies. We note that the regression models are based on a limited set of sites in a small region of the Bay, and observations at many more sites in a diversity of settings are needed to assess their broad applicability.

4.3. Sediment and nutrient storage in the coastal zone

Because there was no obvious difference in subtidal sediments related to SAV presence/absence, we focused on differences between living and reference shorelines rather than potential influences of SAV. Though not statistically significant, the mean sedimentation rate and nutrient concentrations tended to be higher (mean values were 40% higher for rates and ~ 25–35% higher for nutrients; $p > 0.18$ for all) adjacent to reference shorelines compared to those adjacent to living shorelines. This resulted in mean nutrient burial rates (product of sedimentation rate and nutrient concentration) that also tended to be higher adjacent to reference shorelines. C burial was 66% higher ($p = 0.20$), N burial was 67% higher ($p = 0.21$), and P burial was 75% higher ($p = 0.11$). While these differences were not statistically significant, they do suggest increased burial adjacent to reference shorelines. Mean values were highly influenced by anomalously high sedimentation rates and nutrient concentrations adjacent to both the living and reference shoreline at QL. When QL was removed from the dataset, mean values decreased dramatically, but the disparity between the means actually increased: C burial was 90% higher ($p = 0.27$), N burial was 88% higher ($p = 0.30$), and P burial was 82% higher ($p = 0.20$). This highlights the potential for observations at a single site to unduly influence mean values with our small sample size. Quantifying nutrient burial rates at a broad suite of sites is an area ripe for future research.

The differences in nutrient burial between living and reference shorelines could be linked to the differences in shoreline erosion, which affects sediment supply to the subtidal. They could also be explained by differences in where sediment is stored in the coastal zone. Living shorelines have additional intertidal habitat in which to store sediment and nutrients (reference shorelines typically had eroding banks that limit intertidal extent). Subtidal burial rates were lower adjacent to living shorelines than reference shorelines, but living shorelines have additional burial in the marsh, resulting in higher burial across the coastal zone. While the high variability and small number of sites again obscure statistics, this suggests that the locus of sedimentation is shifted to the intertidal environment in areas with living shorelines. Marsh vegetation plays a key role in sediment retention, with sedimentation rates being positively correlated with stem density (except at QL) similar to observations in natural marshes. One implication is that living shorelines with denser vegetation are more effective at trapping sediments and thus more likely to be sustainable with respect to RSLR. In that case, changes in stem density as living shorelines age, from initial sparse plantings to fully developed marshes, could be used to predict the future trajectory of the living shoreline. Another implication is that models of processes in living shorelines need to account for potential changes as living shorelines mature.

Nutrient concentrations were also quite high in marsh sediments. Sediment PN and PC concentrations were significantly higher in the marshes than in the subtidal adjacent to living shorelines and reference shorelines ($p = 0.04$ for all). PP was statistically similar ($p = 0.12$ for living shorelines; $p = 0.17$ for reference shorelines), but the mean PP concentration was 72% and 57% higher in the marshes than in the subtidal adjacent to living shorelines and reference shorelines, respectively. Higher nutrient concentrations likely reflect contributions from the vegetation to marsh sediments, which is also evident in the much higher organic content of marsh sediments. Because PC, PN, and sedimentation rates were all much higher in the marshes of living shorelines than in the adjacent subtidal, PC and PN burial rates in marshes were significantly higher than in the adjacent subtidal. While PP burial rates were not statistically different among the three environments, the mean PP burial rate in the marshes was an order of magnitude higher than adjacent to living shorelines. Summing mean values across the coastal zone, areas with living shorelines store 33.3 mg/cm^2/y PC, 3.0 mg/cm^2/y PN, and 10.0 mg/cm^2/y PP. This can be compared with storage at shorelines with eroding banks (and thus negligible intertidal) of 13.4 mg/cm^2/y PC, 1.3 mg/cm^2/y PN, and 4.2 mg/cm^2/y PP. Thus, storage with living shorelines is $\sim 3\times$ that of eroding shorelines due to the additional storage in marshes.

As mentioned previously, our data represent summertime conditions, when sedimentation should be enhanced in both environments due to the presence of vegetation in the marsh and some subtidal sites, as well as the lack of energetic storm events. In reality, complex feedbacks varying over space and time control sediment exchange between the intertidal and subtidal (Donatelli et al., 2018; Vona et al., 2021). Plants die back in both environments in winter. In the marsh, residual plant litter can help retain and even trap sediment (e.g., Elmore et al., 2016), but SAV usually disappears completely and leaves sediment vulnerable to erosion by winter storms (Russ and Palinkas, 2018; Bolton, 2020). This sediment could be transported onto the marsh platform and/or to downstream environments. Storing sediment in marshes rather than supplying them to adjacent waters could help improve water quality and is needed to maintain marsh elevation in the tidal frame. But, this could come at expense of supplying sediment to nourish downstream marshes and/or SAV beds. Although there were no obvious relationships between living shorelines, sediment, and SAV in this study, there are other trade-offs in ecosystem services that must be fully evaluated if marsh creation for living shorelines might directly replace subtidal SAV habitat (e.g., fish versus waterfowl habitat).

5. Conclusions

This study found that living shoreline installation effectively reduces shoreline erosion and usually results in seaward movement of the shoreline from construction. Living shoreline installation does not cause systematic changes to the subtidal habitat in adjacent waters, with post-installation sediment characteristics being closely linked to pre-installation conditions. Insights into post-installation sediment characteristics at individual sites can be gleaned from regression models that combine the historical shoreline-change rate and average SAV bed density before installation, although local geomorphic processes likely play a larger role. Living shoreline installation does not cause systematic changes to SAV distributions. Rather, SAV distributions at individual sites followed regional trends likely driven by water quality. Living shorelines greatly enhance sediment and nutrient burial in the coastal zone due to additional burial in the intertidal. This study identified many areas ripe for future research such as changes as living shorelines mature, impacts on downstream intertidal and subtidal environments, and studies in areas with denser and more persistent SAV, though the latter is limited by current regulations that discourage living shorelines near SAV.

CRediT authorship contribution statement

Cindy M. Palinkas: Conceptualization, Formal analysis, Writing – original draft, Funding acquisition. **Miles C. Bolton:** Investigation, Writing – original draft. **Lorie W. Staver:** Conceptualization, Investigation, Writing – review & editing, Funding acquisition.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Data availability

Data will be made available on request.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.ecoleng.2023.106944.

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