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Influence of Shoreline Stabilization Structures on the Nearshore Sedimentary Environment in Mesohaline Chesapeake Bay

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Abstract Shorelines around many estuaries and coastal embayments are rapidly eroding (approximately several meters/year), with more rapid erosion rates expected in the future due to natural and anthropogenic stressors. In response, a variety of techniques have been used to stabilize shorelines, but there are limited quantitative, long-term data available about their effects on the sedimentary environment immediately adjacent to them (i.e., the nearshore). This study evaluated changes in sediment characteristics (mud and organic content) and accumulation rates associated with installation of breakwaters, riprap, and living shorelines with ("hybrid") and without ("soft") a structural component. ^{210}Pb (half-life 22.3 years) geochronologies were used to identify horizons in core profiles that corresponded to years when structures were built. Sites with naturally eroding shorelines (i.e., no structures) were used as a control group at which any sedimentary changes represent broad environmental trends, in contrast to changes at the protected sites that also include the influence of structures. Observations were placed within the context of modeled wave climate, shoreline-erosion rates, land use, dominant sediment source, and the apparent effect on submersed aquatic vegetation (SAV) inhabiting the nearshore sedimentary environment. The main conclusion of this study is that there was no "one size fits all" answer to anticipated impacts of structures on nearshore sedimentary environments. Instead, specific changes associated with structures depended on individual site characteristics, but could be predicted with multiple linear regression models that included structure type, shoreline-erosion rate, dominant sediment source, and land use. Riprap or breakwater installation had either positive or no obvious impact on SAV at six of seven sites but negatively impacted SAV at one riprapped site. No obvious impacts on SAV were observed at living shoreline sites.

Keywords Breakwater · Living shoreline · Riprap · Submersed aquatic vegetation (SAV) · ^{210}Pb · Shoreline erosion · Sediment accumulation rates · Sedimentary characteristics (mud and organic content)

Introduction

Estuaries and coastal embayments are increasingly impacted by coastal erosion related to multiple stressors, including increasing rates of sea-level rise (SLR; global acceleration of rates ~ 0.01 mm/y^2; Church and White 2006, Gehrels and Woodworth 2013; Jevrejeva et al. 2008; Woodworth et al. 2009), urbanization (e.g., Allen 2000; Erdle et al. 2008), and storms (Sutton-Grier et al. 2015). Coastal erosion is compounded by losses of submersed aquatic vegetation (SAV) and tidal marsh communities (Day et al. 2008; Orth et al. 2010; Watson et al. 2011) that could provide resilience against these stressors. Erosion leads to property loss, increased levels of suspended sediments and turbidity in the adjacent aquatic environment, as well as degraded water quality (Wells et al. 2003). Shoreline hardening can be an effective preventive measure for coastal erosion (Charlier et al. 2005), and rates of hardening are increasing worldwide, especially on developed and urbanized coastlines (Airoldi et al. 2005; Nordstrom 2003). A recent conservative estimate of armored

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shoreline in the continental US is 14% (Gittman et al. 2014), and many shorelines are already > 50% hardened (Erdle et al. 2008). However, eroding shorelines can also provide important ecosystem services, such as beach habitat and a source of sediment to nourish SAV beds and marshes, that are lost when shorelines are hardened and the land-water connection is severed (Bilkovic and Roggero 2008; Scyphers et al. 2015).

Historically, shoreline stabilization occurred largely through use of "hard" structures, such as bulkheads and seawalls. These structures can provide new ecosystem services, such as rocky habitat for macroalgae (Seitz et al. 2006), but they also can have significant negative ecosystem impacts. For example, bulkheads typically reflect incident waves, leading to increased sediment scour and deepening of nearshore waters, resulting in degraded benthic habitats and loss of diversity (Currin et al. 2010 and references therein). Further, there are often trade-offs between positive and negative impacts—e.g., breakwaters reduce energy in their landward portion, benefitting SAV, but trap fine and organic particles that can degrade SAV habitat (Currin et al. 2010). More recent efforts have focused on natural alternatives, such as oyster reefs and marshes (Scyphers et al. 2011; Sutton-Grier et al. 2015). In particular, "living shorelines" (i.e., narrow bands of marsh habitat with or without additional structures; Burke et al. 2005) have gained traction in the management community, resulting in regulatory efforts to encourage their use in at least four states (North Carolina, Maryland, Virginia, and Delaware; Currin et al. 2010). Living shorelines preserve the land-water connection and provide many of the ecosystem services attributed to natural marshes (e.g., sediment and nutrient retention and wave attenuation; Davis et al. 2015; Manis et al. 2015). However, they are also subject to the same stressors as natural marshes, and questions remain regarding their potential for long-term survival.

In Chesapeake Bay, local SLR is especially rapid (~3–4 times the global average; Boon 2012; Sallenger et al. 2012) and approximately one third of the Bay's shoreline is classified as eroding (Chesapeake Bay Program 2006). Approximately 25% of the Chesapeake Bay shoreline (main stem and tributaries) is hardened, and some sub-watersheds are > 50% armored (Patrick et al. 2014). The incidence of eroding shorelines is even higher in the Maryland portion of the Bay, where ~ 70% of shorelines are classified as eroding (Hennessee et al. 2002, 2003a). Recent legislation in Maryland specifically promotes installation of living shorelines where appropriate (http://www.mde.state.md.us/programs/Water/WetlandsandWaterways/Pages/LivingShorelines.aspx), but a wide variety of shoreline stabilization techniques are used in the Maryland Chesapeake Bay, ranging from complete hardening (stone revetment, also called riprap, and bulkheading) through emergent offshore structures (breakwaters, groins, and tombolas) and submerged breakwaters to living shorelines. While the qualitative effects of these stabilization techniques are known in general, more detailed information is often limited due to lack of baseline data prior to structure installation and/or long-term monitoring data after installation. Structure influence on nearshore benthic habitat is especially important for SAV, which are a keystone species in Chesapeake Bay (Batiuk et al. 2000).

The focus of this study is a comparative synthesis of observed changes in the nearshore sedimentary environment and SAV abundances associated with shoreline stabilization structures in mesohaline Chesapeake Bay (see Fig. 1). Specifically, we aim to: (1) identify structure influence in sediment cores and characterize pre- and post-construction sediment characteristics and accumulation rates; (2) relate these observations to current and historical SAV abundances; and (3) integrate data within the context of physical setting (dominant sediment source, land use, shoreline type, wave climate). In doing so, we create conceptual models of structure influence that are relevant to resource managers and provide a basis for future studies. We hypothesize that structural influence in the nearshore depends on multiple factors such as structure type, wave climate, dominant sediment source, and adjacent land use. Factors that increase fine and organic content of bottom sediments after construction (e.g., decrease in wave energy and/or supply of riverine fines to areas landward of breakwaters) are likely to be detrimental to SAV. In contrast, factors that increase coarse (sand) accumulation (e.g., sand emplacement during construction) are likely to benefit SAV.

Methods

Site Selection

Sites were selected from previous studies examining specific aspects of shore erosion and shoreline stabilization structures around Chesapeake Bay. Four types of shorelines were considered: (1) offshore segmented breakwaters ($n = 4$; Palinkas et al. 2016), (2) riprapped shorelines ($n = 3$), (3) both soft (vegetation only; $n = 3$) and hybrid (structure and vegetation; $n = 3$) living shorelines (Burke et al. 2005), and (4) naturally eroding shorelines ($n = 4$; Hill et al. 2003). All sites were in the mesohaline salinity zone, except for Elk Neck and Meeks Point in the oligohaline salinity zone (Fig. 1); structure ages (time since installation) ranged from 3 to 32 years (Table 1).

Sediment Analysis Methods

One vibracore (~ 3-m long) was collected in the nearshore (~ 1–2 m water depth) adjacent to each site. Three companion push cores (5-cm diameter, ~ 20-cm long) were taken near each vibracore to capture relatively undisturbed surface sediment and to assess small-scale variability. Vibracores were

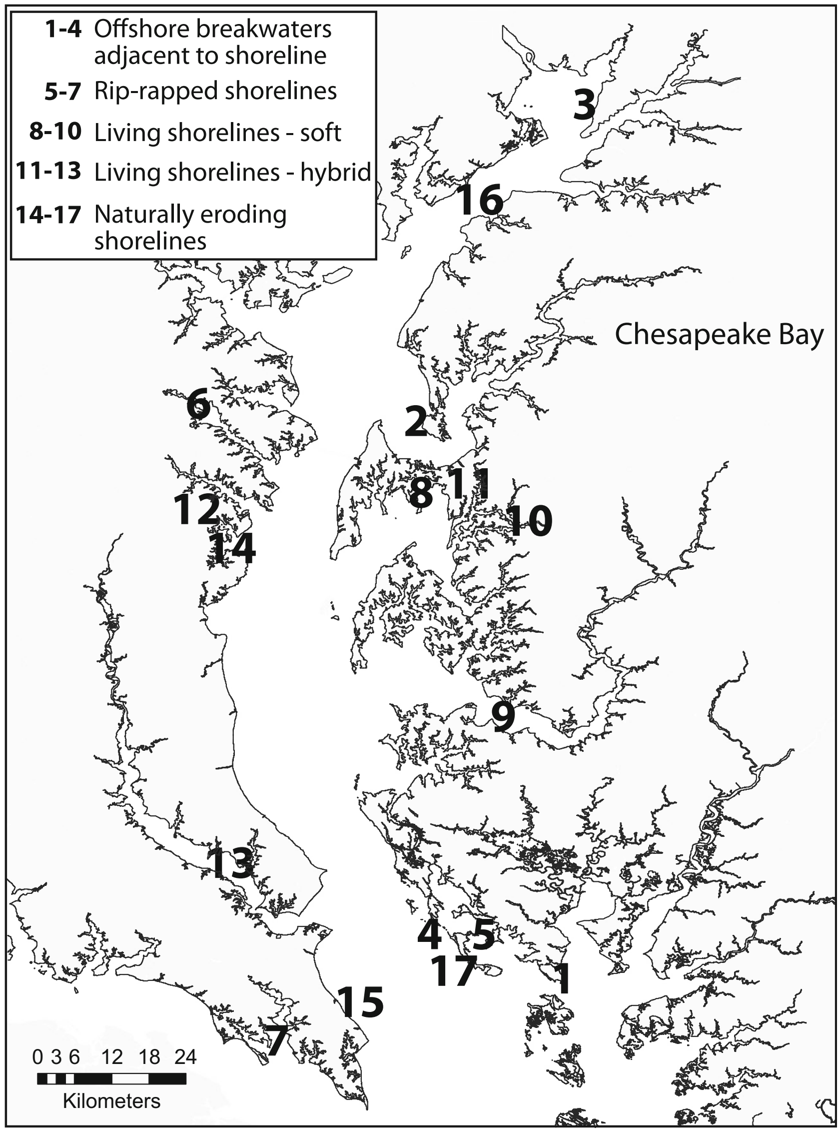

Fig. 1 Map of study sites in the Maryland portion of Chesapeake Bay. Site numbers on map correspond those in Table 1 and are grouped by category

returned to the laboratory and frozen in a vertical position until further analysis. Push cores were sectioned into 1-cm increments immediately upon returning to the lab. Prior to analysis, frozen vibracores were thawed, cut in half lengthwise, and sectioned into 1- (upper 20 cm) and 2-cm (rest of the core) increments. Sediments were then analyzed for grain size, organic matter, and the presence of the naturally occurring radioisotope ^{210}Pb (half-life 22.3 y). Sediment cores were collected in 2012 and 2013, except for breakwater sites, which were sampled in 2008 and 2009.

For each 1- or 2-cm increment analyzed, samples were processed for grain size by wet-sieving at 64 μm to separate the mud-sized (< 64 μm) and sand-sized (> 64 μm) components. The mud-sized component was subsequently analyzed with a Sedigraph 5120 (Coakley and Syvitski 1991) and dried. The sand-sized component was dry sieved from 64 to 500 μm, using a standard set of 13 sieves. Mud and sand data were joined to calculate the median diameter. Organic content was determined via combustion at 450 °C for 4 h (Erftemeijer and Koch 2001).

^{210}Pb (half-life 22.3 y) is a useful geochronometer of sedimentation processes occurring over the last ~ 100 y (~ 4–5 half lives, typically assumed as the detection limit). ^{210}Pb is produced by the decay of ^{238}U and is supplied by precipitation, runoff, and decay of its effective parent ^{226}Ra (Nittrouer et al. 1979). It has been used in nearshore Chesapeake Bay environments previously (Palinkas et al. 2010; Palinkas and Koch 2012). ^{210}Pb activities were measured via alpha spectroscopy (Canberra Alpha Analyst), using the methods of (Palinkas and Nittrouer 2007). Samples for all radioisotopic analyses were counted for ~ 24 h and decay-corrected to the time of sample collection. Because ^{210}Pb is preferentially adsorbed to fine particles (Andersen et al. 2011; Nittrouer et al. 1979), activities were normalized to the mud content (activity divided by percent mud) to remove the effects of grain-size variations.

Table 1 Basic characteristics of sites grouped by category (estimated total length in structure in parentheses); site numbers correspond to those in Fig. 1. Surficial mud and organic content values represent the average over the uppermost 10 cm of 3 push cores. "Dominant species" refers to submersed aquatic vegetation (SAV) present during field surveys

Site	Category	Install date	Age at sampling	Surficial mud, %	Surficial organic, %	Dominant species
Bishops Head	Breakwater (190 m)	1996	11	51.3	4.1	*Ruppia maritima*
Eastern Neck	Breakwater (365 m)	1992	15	2.9	2.2	*Zannichellia palustris*
Elk Neck	Breakwater (255 m)	2005	3	17.6	1.4	*Hydrilla verticillata*
Hoopers Island	Breakwater (1160 m)	1994	14	17.4	2.1	*Zostera marina*
Average				**22.3 ± 20.5**	**2.5 ± 1.6**	
Honga	Riprap (530 m)	1997	15	22.1	2.0	*Ruppia maritima*
Severn	Riprap (205 m)	2001	12	4.9	1.2	NA
St. Marys	Riprap (70 m)	2003	9	28.7	1.9	NA
Average				**18.6 ± 12.3**	**1.7 ± 0.4**	
CBEC	Soft living (150 m)	2002	11	3.6	0.7	NA
Horn Point	Soft living (105 m)	1980	32	59.8	4.6	NA
Wye	Soft living (200 m)	1990	22	3.9	0.8	NA
Average				**22.4 ± 32.4**	**2.0 ± 2.2**	
Aspen	Hybrid living (200 m)	1995	17	3.8	0.7	NA
Camp Letts	Hybrid living (290 m)	2007	6	8.9	1.3	NA
Patterson Park	Hybrid living (140 m)	1998	15	2.8	1.2	NA
Average				**5.1 ± 3.3**	**0.9 ± 0.3**	
Cheston Point	Naturally eroding	NA	NA	4.6	1.6	NA
Elms	Naturally eroding	NA	NA	5.5	0.7	NA
Meeks Point	Naturally eroding	NA	NA	2.9	0.7	NA
Richland Point	Naturally eroding	NA	NA	5.4	1.0	NA
Average				**4.6 ± 1.2**	**1.0 ± 0.4**	

Average values for each shoreline type are given in bolded text and listed in the row following the individual sites

Depth-integrated ^{210}Pb inventories were then calculated for each core and used to calculate sediment accumulation rates using the constant initial concentration (CIC) model (Appleby and Oldfield 1978). This model does not require an assumption of steady sedimentation; rather, the age (t) of sediments at depth z is:

$$t = \frac{1}{\lambda} ln\left(\frac{A_0}{A_z}\right) \quad (1)$$

where A_z is the cumulative inventory of excess ^{210}Pb activity beneath depth z, and A_0 is the cumulative inventory of excess ^{210}Pb activity in the sediment column. Excess ^{210}Pb activities were taken from the base of cores, where activities have a low, uniform value. Sediment ages were used to identify horizons in down-core profiles that correspond to years when structures were built; down-core sedimentation rates were calculated by dividing the sediment age by its depth. "Post-construction" sediments thus resided above the horizon corresponding to structure installation. "Pre-construction" sediments were identified as the portion of cores below the structure installation horizon. For example, the living shoreline at CBEC was 11 years old at the time of sampling in 2013; thus, "post-construction" sediments were deposited from 2002 to 2013, and "pre-construction" sediments were represented by those deposited from 1991 to 2002. This approach minimized potential inclusion of long-term historical changes in sediment character present at the base of some cores. Potential temporal changes at naturally eroding sites were assessed by considering the 14 years (average age of structures at other sites) prior to the study (1998–2012) as "recent" conditions; "past" conditions were represented by the 14 years preceding the "recent" time period (1984–1998). These data provide context for interpreting any observed changes at the protected sites; i.e., any changes at the eroding sites reflect broad environmental trends, whereas changes at the protected sites also include structural influences.

Vegetation Methods

Vegetation presence/absence and species composition at each site was assessed along three transects from the shoreline to a depth of 1.5 m (limiting depth for SAV in Chesapeake Bay; Kemp et al. 2004) coincident with sediment core collection—i.e., in 2012 or 2013 for all sites except breakwater sites, which were sampled in 2008 or 2009. This means that transects were located seaward of riprap and living shoreline structures (co-located with the shoreline) and landward of breakwaters (water depths seaward of breakwaters were > 1.5 m). "Dominant species" in Table 1 refers to the dominant SAV species present in these surveys. In addition, SAV distributions before and after structure installation were assessed via aerial images taken annually within Chesapeake Bay since 1980 (annual reports from the Virginia Institute of Marine Science (VIMS); e.g., Orth et al. 1998). Aerial photos for each site were obtained from 3 years prior to construction to 2013. Aerial photos were georeferenced using ArcMap, and the area influenced by the structure was quantified. To separate structure effects on SAV distribution from inter-annual fluctuations in regional SAV distribution, the time series of SAV distribution created for each site was compared with the corresponding SAV time series for the broader segment (as defined by the Chesapeake Bay Program (CBP); Batiuk et al. 2000) of the Chesapeake Bay in which the site was located.

Wave Modeling and Ancillary Data

Climatological wind-wave estimates were determined for each site over a 21-year time period (1985–2005), following Sanford and Gao (2017). The approach provided better correspondence of the wind-wave climatologies to historical shoreline-erosion rates, calculated by differencing over comparable time periods. Climatologies were produced with Simulating Waves Nearshore (SWAN; Booij et al. 1999; Ris et al. 1999), using the identical model grid and bathymetry as the Army Corps of Engineers for the US EPA Chesapeake Bay Program (CBP) hydrodynamic and water-quality models (Cerco and Noel 2004). Grid resolution was approximately 1 km in the axial direction and 0.4 km in the lateral direction; the calculation time step was 10 min. Wind data were linearly interpolated between the two nearest stations onto over-water model grid points, based on five long-term wind stations located around Chesapeake Bay. Fetch calculations were not available from SWAN and were instead calculated using the CBP wave model. The CBP wave model is a parametric wind-wave model based the formulations of Young and Verhagen (1996) and applied to Chesapeake Bay (Harris et al. 2012). Hourly water levels for the entire Chesapeake Bay from 1985 to 2005 were provided by the Waterways Experiment Station (CH3D-WES) model (Johnson et al. 1993). Model predictions were carefully calibrated to observed water levels at NOAA tide gauges by Johnson et al. (1993). The model predictions provide much higher resolution water level time series, directly matched to the wave model grid. Specific model outputs used in this study were significant wave height, the top 5% of significant wave heights, peak wave period, bottom-orbital velocity, water level, tidal range, and fetch.

Shoreline-erosion rates were determined with the Maryland Coastal Atlas (http://dnr2.maryland.gov/ccs/Pages/coastalatlas.aspx). These rates were calculated from digitized shorelines from 1841 to 1995 input into the Digital Shoreline Analysis System (DSAS; Danforth and Thieler 1992) by the Maryland Geological Survey and partners, producing nearly 250,000 shore-normal transects with associated rates of change along the Atlantic coast, coastal bays, and Chesapeake Bay and its tributaries (Hennessee et al. 2002, 2003a, b). Transects are spaced ~ 20 m apart (smaller than the shortest estimated structure length; see Table 1); the shoreline-erosion rate from the unprotected shoreline transect nearest each core location was used.

Shoreline inventories from the Center for Coastal Resource Management (VIMS; http://ccrm.vims.edu/gis_data_maps/shoreline_inventories/) provided data on land use, shoreline type, and bank heights. These data are available for every county in Maryland; surveys occurred in the early to mid-2000s and not necessarily coincident (e.g., data for Severn and Camp Letts in Anne Arundel County were collected in 2005; data for Honga, Bishops Head, and Hoopers Island in Dorchester County were collected in 2003; Berman et al. 2006 and 2003, respectively). Data were imported into ArcMap; data corresponding to the shoreline segment nearest each site were used. Likely sediment sources were inferred from proximity to major tributaries of Chesapeake Bay (e.g., Elk Neck), obvious changes in down-core profiles implying sand emplacement (e.g., Hoopers Island), or proximity to shoreline structures themselves (e.g., marsh for living shorelines, riprap for riprapped sites).

Statistical Analyses

All statistical analyses were performed using R statistical software. T-tests were used to compare pre- and post-construction conditions at each site, wave climate conditions across different structure types, and changes in sedimentary conditions among different land uses and different sediment sources. T-tests also were used to compare average surficial sediment characteristics, as well as average pre- and post-construction conditions, among structure types. Relationships between various site properties and changes in sedimentary conditions were explored with multiple linear regression models that included only sites with structures. Response and explanatory variables were chosen based on potential relevance to resource management—e.g., managers are most likely to be interested in predicting changes in sediment character and accumulation rates based on site properties. Similarly, site properties that can be inferred (e.g., sediment source) or obtained from publicly available data (e.g., shoreline-erosion rate) are likely most relevant. Thus, three separate models were constructed to predict changes in mud content, changes in organic content, and changes in sedimentation rates. Explanatory variables included structure type and age, shoreline-erosion rate, sediment source, and land use. If the shoreline was identified as the likely sediment source, it was subcategorized according to type (i.e., marsh or riprap). To reduce the number of land-use categories, residential, commercial, and paved land uses were combined into a single "developed" category; forest and scrub-shrub land uses were combined into a single "vegetated" category. Initial regression models included all variables; variables were removed stepwise to obtain the most parsimonious result.

For all statistical analyses, $p < 0.1$ was considered significant, rather than the more rigorous requirement of $p < 0.05$ to lend insight into physical differences that might otherwise be excluded due to relatively low sample size. Also for this reason, general trends are described for differences that are not statistically significant.

Results

Patterns of Sedimentation and Relationship with SAV Cover

Characteristics of surficial (averaged over the topmost 10 cm of push cores) sediments varied spatially among sites (Table 1). Mud content ranged from 2.9% at Meeks Point and Eastern Neck to 59.8% at Horn Point, and organic content ranged from 0.4% at Eastern Neck to 4.9% at Bishops Head. On average, sediments tended to be finest (highest mud content) at soft living shoreline sites and breakwater sites, and sediments tended to be coarsest (lowest mud content) at hybrid living shorelines and naturally eroding sites. None of the differences in average mud content were significant, however. Correspondingly, sediments at soft living shorelines and breakwater sites had the highest organic content, whereas sediments at hybrid living shorelines and naturally eroding sites had the lowest organic content. Average organic content at breakwater sites was significantly higher than for hybrid living shorelines ($p = 0.09$) and eroding shorelines ($p = 0.08$).

Temporally, and averaged across all sites within each category (Table 2), sediments were coarser (less mud) after structure installation for all categories and less organic after structure installation for all categories except breakwaters, for which organic content increased. Average sedimentation rates decreased slightly for breakwaters, soft living shorelines, and naturally eroding sites; remained similar for riprapped shorelines; and increased slightly for hybrid living shorelines. However, none of these general trends were statistically significant. No consistent relationships were observed between structure age and changes in mud/organic content or sedimentation rates (Tables 1 and 2). After structure installation, SAV coverage increased at two of the four breakwater sites, increased at one of the three riprap sites, decreased at one of the three riprap sites, and did not change at the remaining two breakwater sites and one riprap site (Table 3).

Sediments at three of the four breakwater sites became sandier (decreased mud content) after construction, except at Elk Neck where sediments became significantly muddier and more organic after construction. Note that Elk Neck was by far the youngest breakwater installation (Table 1), and it was near the mouth of the Susquehanna River, a major source of fine sediments. At Elk Neck, SAV (*Vallisneria americana* and *Hydrilla verticillata*) were historically absent but colonized the landward portion of the breakwater coincident with construction and followed the general trends of the CBP segment thereafter (Table 3). At Eastern Neck, mud and organic content decreased after construction, with no obvious influence on SAV (*Zannichellia palustris* present pre- and post-installation, Table 3). Mud and organic content also decreased at Hoopers Island (Table 2 and Fig. 2). Here, SAV (*Ruppia maritima* and *Zostera marina*) colonized the landward portion of the breakwater 5 years after construction. Lastly, while mud content decreased following construction at Bishops Head, organic content increased, with no obvious influence on SAV (*Ruppia maritima*).

Sediments at all riprapped sites became coarser and less organic after installation. At Honga (Fig. 3), sedimentation rates also decreased, with no obvious impact on SAV (*Ruppia maritima*). Sedimentation rates also decreased at St. Marys; however, SAV colonized the site coincident with construction and subsequently followed the temporal trend of the CBP segment (SAV was absent from the site and segment in 2012). Sedimentation rates increased at Severn; temporal patterns of SAV coverage followed that of the CBP segment until 5 years post-construction, then SAV disappeared from the site but remained present within the segment.

Mud and organic content of sediments decreased at all sites with soft living shorelines. Sedimentation rates decreased at CBEC but remained unchanged at Wye and Horn Point. SAV was absent from all of these sites during the survey and for the majority of time in the respective historical records. SAV was also absent from all hybrid living shoreline sites. Sedimentation rate changes at hybrid living shoreline sites were equivocal; they decreased at Patterson Park (Fig. 4) but increased at Severn and Camp Letts.

For naturally eroding sites, mud and organic content decreased significantly at Elms between the "before" (1984–1998) and "after" (1998–2012) time periods but remained similar at Richland Point and Meeks Point. Average sedimentation rates decreased at all naturally eroding sites. Note that bottom sediments at Cheston Point were erosional (no excess ^{210}Pb activity) throughout the time period considered, which precluded identification of "recent" or "past" depth intervals. SAV has not been present at any of these sites since 1980, although it has been persistent in the corresponding CBP segments.

Table 2 Sediment characteristics (mud and organic content) and accumulation rates before and after structure installation at each site. For naturally eroding shorelines, "before" and "after" refer to 1984–1998 and 1998–2012, respectively (see Methods). Asterisks (*) in the "after" column denote significant changes ($p < 0.1$; t-tests) from the "before" conditions; corresponding p values are given. Bolded values in the "after" column indicate decreased values from the "before" condition

Site	Category	% Mud before	% Mud after	% Organic before	% Organic after	Rate before, cm/y	Rate after, cm/y
Bishops Head	Breakwater	61.4	**48.7*** ($p = 0.004$)	1.9	5.2* ($p < 0.001$)	3.9	**1.3*** ($p = 0.002$)
Eastern Neck	Breakwater	62.1	**43.8**	2.9	**2.2**	6.3	8.5
Elk Neck	Breakwater	6.6	18.3* ($p = 0.004$)	0.5	1.5* ($p = 0.01$)	0.6	**0.4**
Hoopers Island	Breakwater	74.8	**37.6**	3.4	**2.4*** ($p = 0.08$)	1.7	**1.6**
	Average	**51.2 ± 30.4**	**37.1 ± 13.3**	**2.2 ± 1.3**	**2.8 ± 1.7**	**3.1 ± 2.5**	**2.9 ± 3.7**
Honga	Riprap	29.1	**6.2*** ($p = 0.006$)	1.3	**0.9**	1.9	**1.0*** ($p = 0.005$)
Severn	Riprap	9.3	**6.4*** ($p = 0.008$)	2.1	**1.4**	2.4	3.5
St. Marys	Riprap	85.4	**25.1*** ($p < 0.001$)	3.2	**1.5**	0.5	**0.3*** ($p = 0.05$)
	Average	**41.3 ± 39.5**	**12.6 ± 10.9**	**2.2 ± 0.9**	**1.3 ± 0.3**	**1.6 ± 1.0**	**1.6 ± 1.7**
CBEC	Soft living	38.8	**4.0*** ($p = 0.003$)	1.9	**0.7*** ($p = 0.001$)	3.4	**1.2*** ($p < 0.001$)
Horn Point	Soft living	84.2	**56.2*** ($p = 0.001$)	4.8	**4.1**	0.7	0.7
Wye	Soft living	4.8	**3.2**	0.7	**0.6**	0.8	0.8
	Average	**42.6 ± 39.8**	**21.1 ± 30.4**	**2.5 ± 2.1**	**1.8 ± 2.0**	**1.6 ± 1.5**	**0.9 ± 0.3**
Aspen	Hybrid living	9.6	18.8	0.7	1.7	1.4	1.7* ($p = 0.03$)
Camp Letts	Hybrid living	20.4	**8.1*** ($p = 0.06$)	1.9	**1.1*** ($p = 0.004$)	3.7	4.4* ($p = 0.06$)
Patterson Park	Hybrid living	3.4	**1.5*** ($p = 0.009$)	1.0	**0.5**	0.3	**0.2*** ($p = 0.03$)
	Average	**11.1 ± 8.6**	**9.5 ± 8.7**	**1.2 ± 0.6**	**1.1 ± 0.6**	**1.8 ± 1.7**	**2.1 ± 2.1**
Cheston Point	Naturally eroding	NA	NA	NA	NA	NA	NA
Elms	Naturally eroding	85.2	**55.5*** ($p = 0.02$)	4.6	**4.0**	0.9	**0.7**
Meeks Point	Naturally eroding	3.7	**3.5**	0.4	0.4	2.7	**1.4*** ($p = 0.005$)
Richland Point	Naturally eroding	5.9	**5.8**	0.8	0.8	0.7	**0.5**
	Average	**31.6 ± 46.5**	**21.6 ± 29.4**	**1.9 ± 2.3**	**1.7 ± 1.9**	**1.4 ± 1.1**	**0.9 ± 0.5**

Wave Climate, Land Use, and Sediment-Source Effects

Wave climate data were examined to see whether there were differences between characteristic wave forcing factors among the shoreline types. Wave characteristics were averaged by structure category, combining soft and hybrid living shorelines into a single "living shoreline" category because of data limitations. All variables considered were similar among structure categories, except for fetch, bottom-orbital velocity (Fig. 5), and shoreline-erosion rate (Fig. 5 and Table 3). These quantities tended to be higher for breakwaters and naturally eroding shorelines than for riprap or living shorelines. However, differences were statistically significant only between breakwaters and living shorelines and only for fetch ($p = 0.09$).

Land uses from the VIMS shoreline inventory were grouped into three categories: developed (residential, commercial, paved, bare), agricultural, and vegetated (forest, scrub-shrub) (Table 3). The most common land use was development, which occurred at three of the four breakwater sites, two of the three riprapped sites, one of three soft living shorelines, and two of three hybrid living shorelines. Agricultural land uses occurred at one breakwater site, one riprapped site, and one soft living shoreline. Vegetated land uses occurred only at one soft living shoreline, one hybrid living shoreline, and all the naturally eroding sites. Dominant sediment sources, inferred as described in the Methods, were most often from shorelines. All living shorelines, by design, had marshes along their shorelines; similarly, riprapped sites had riprap along their shorelines. Naturally eroding shorelines all had beaches along their shorelines. There were two other sediment sources present at study sites—sand emplaced during construction (Eastern Neck, Hoopers Island, and St. Marys), and riverine supply of fine sediments (Elk Neck). Breakwater sites had the greatest diversity of dominant sediment sources, whereas all living shorelines and naturally eroding sites were dominated by local shoreline sources.

The influences of land use on nearshore sedimentary environments were evaluated by comparing observed changes in

Table 3 Dominant sediment source, shoreline-erosion rate, and land use for each site, as well as an assessment of structure installation appeared to benefit submersed aquatic vegetation (SAV)

Site	Category	Sediment source	Erosion rate (m/y)	Land use	Benefit SAV
Bishops Head	Breakwater	Shoreline - marsh	0.13	Developed (residential)	NA
Eastern Neck	Breakwater	Sand	1.54	Agriculture	NA
Elk Neck	Breakwater	River	0.01	Developed (bare)	Yes
Hoopers Island	Breakwater	Sand	1.86	Developed (paved)	Yes
Honga	Riprap	Shoreline - riprap	0.69	Agriculture	NA
Severn	Riprap	Shoreline - riprap	0.58	Developed (residential)	No
St. Marys	Riprap	Sand	0.25	Developed (residential)	Yes
CBEC	Soft living	Shoreline -marsh	0.00	Developed (residential)	NA
HPL	Soft living	Shoreline - marsh	0.02	Agriculture	NA
Wye	Soft living	Shoreline - marsh	0.66	Vegetated (forest)	NA
Aspen	Hybrid living	Shoreline - marsh	0.13	Developed (residential)	NA
Camp Letts	Hybrid living	Shoreline - marsh	0.39	Developed (commercial)	NA
Patterson Park	Hybrid living	Shoreline - marsh	0.20	Vegetated (forest)	NA
Cheston Point	Naturally eroding	Shoreline - beach	0.77	Vegetated (forest)	NA
Elms	Naturally eroding	Shoreline - beach	0.56	Vegetated (forest)	NA
Meeks Point	Naturally eroding	Shoreline - beach	0.82	Vegetated (forest)	NA
Richland Point	Naturally eroding	Shoreline - beach	2.91	Vegetated (scrub-shrub)	NA

the sedimentary environment (mud and organic content, sedimentation rate) with data on land use for each site (Table 3). Structurally modified sites were examined separately from naturally eroding sites in these analyses, as the latter represent a non-structural "control" group (and they were all vegetated). When sites were grouped by land use, mud content, organic content, and sedimentation rate all decreased post-installation (Fig. 6a). The average decrease in mud content at structural sites was larger for developed and agricultural land uses than for vegetated sites. Conversely, the average decrease in organic content at structural sites was largest for vegetated sites. For naturally eroding sites, the average decreases in mud and organic content were smaller than for any other land-use group. Average decreases in sedimentation rate were similar across all land use groupings. None of the changes shown in Fig. 6a were statistically distinct, given the low number of sites and relatively wide variability.

When sites were grouped by dominant sediment source (Fig. 6b), all average changes were negative except for an increase in mud and organic content for the riverine source at Elk Neck. The greatest average change in mud content at structural sites was a large decrease for shoreline sediment

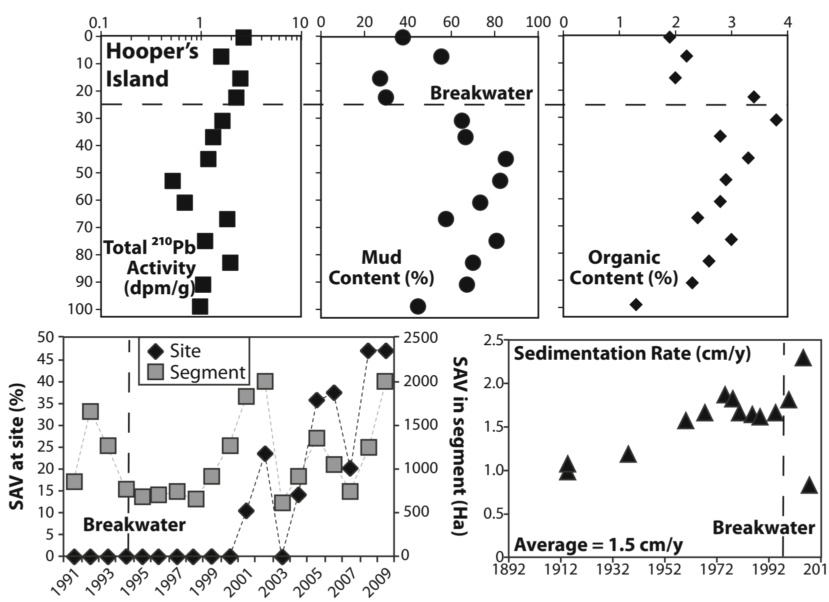

Fig. 2 (Upper; left to right): Down-core profiles of total ^{210}Pb activity, mud content, and organic content for the breakwater site at Hooper's Island. (Lower, left): SAV abundance (hectares) in the Chesapeake Bay Program monitoring segment containing the breakwater site (gray boxes), and SAV cover (percent) observed at the site in aerial photos. (Lower, right): Sedimentation rates versus years calculated with the age-depth model of Appleby and Oldfield (1978). In all figures, the dashed line indicates horizons associated with breakwater installation

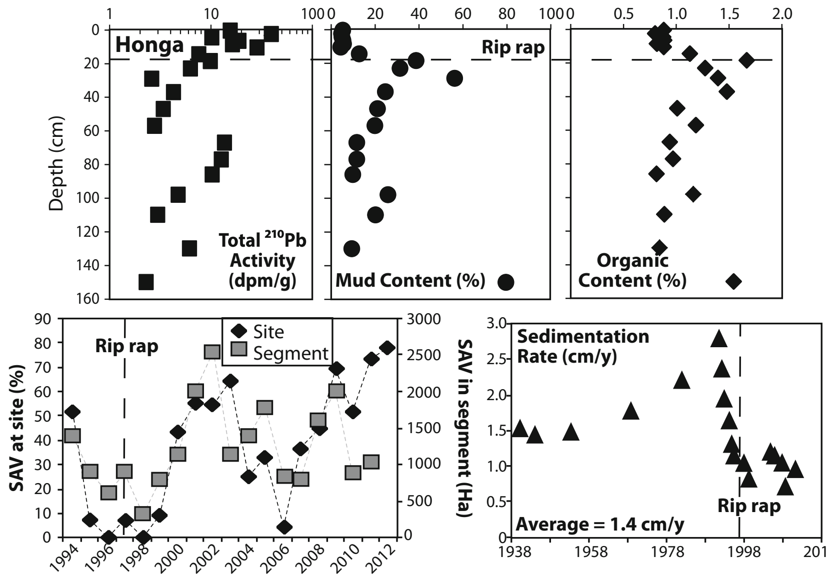

Fig. 3 (Upper; left to right): Down-core profiles of total ^{210}Pb activity, mud content, and organic content for the riprapped site at Honga. (Lower, left): SAV abundance (hectares) in the Chesapeake Bay Program monitoring segment containing the riprapped site (gray boxes), and SAV cover (percent) observed at the site in aerial photos. (Lower, right): Sedimentation rates versus years calculated with the age-depth model of Appleby and Oldfield (1978). In all figures, the dashed line indicates horizons associated with riprap installation

sources, especially at riprapped sites. The greatest average change in organic content at structural sites was the increase for riverine sources, while the least average change in organic content was for shoreline sources, especially marshy shorelines. The greatest average change in sedimentation rates at structural sites was a decrease for shoreline sources, especially marshy shorelines, and the least change was for sand emplacement. For naturally eroding sites, the average change in mud was smaller than for any other sediment-source group, while the average change in organic content was similar to structural sites with marshy shorelines. The decrease in sedimentation rate at naturally eroding sites was greater than for any other sediment-source group. None of the groups shown in Fig. 6b were statistically distinct, given the low number of sites and relatively wide variability.

Inter-Relationships Among Observations

Three separate multiple linear regression models were constructed to predict the average change in mud content, organic content, and sedimentation rates associated with structural sites (Table 4). Note that hybrid living and soft living shorelines were combined into a single "living shoreline" category for these models, and models included only sites with structures. Explanatory variables included structure type and age, shoreline-erosion rate, sediment source, and land use. The best (lowest p value) model for change in mud content was not statistically significant (p = 0.32, adjusted R^2 = 0.48) and included all variables; in this model, structure type and shoreline-erosion rate did have significant ($p < 0.1$) individual coefficients. The best model for change in organic content was

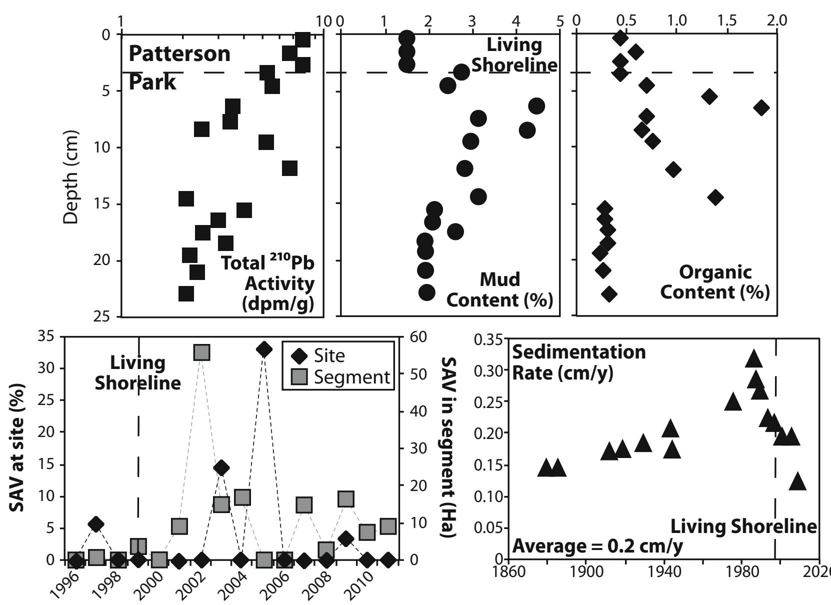

Fig. 4 (Upper; left to right): Down-core profiles of total ^{210}Pb activity, mud content, and organic content for the living shoreline site at Patterson Park. (Lower, left): SAV abundance (hectares) in the Chesapeake Bay Program monitoring segment containing the living shoreline site (gray boxes), and SAV cover (percent) observed at the site in aerial photos. (Lower, right): Sedimentation rates versus years calculated with the age-depth model of Appleby and Oldfield (1978). In all figures, the dashed line indicates horizons associated with living shoreline installation

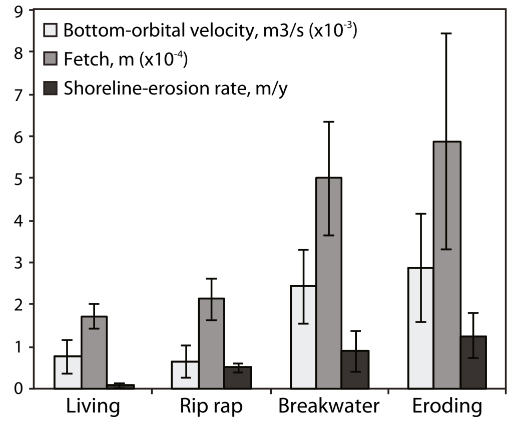

Fig. 5 Average wave climate data for structure categories (errors bars indicate 1 standard deviation). Sites with hybrid and soft living shorelines were combined into a single "living shoreline" category

significant (p = 0.04, adjusted R^2 = 0.63) and included structure type, shoreline-erosion rate, and land use; all coefficients except for vegetated land use were significant at the $p < 0.05$ level. The significant model for change in sedimentation rate with the fewest variables (p = 0.09, adjusted R^2 = 0.38) included sediment source and shoreline-erosion rate; all coefficients were significant at the $p < 0.1$ level. There was a

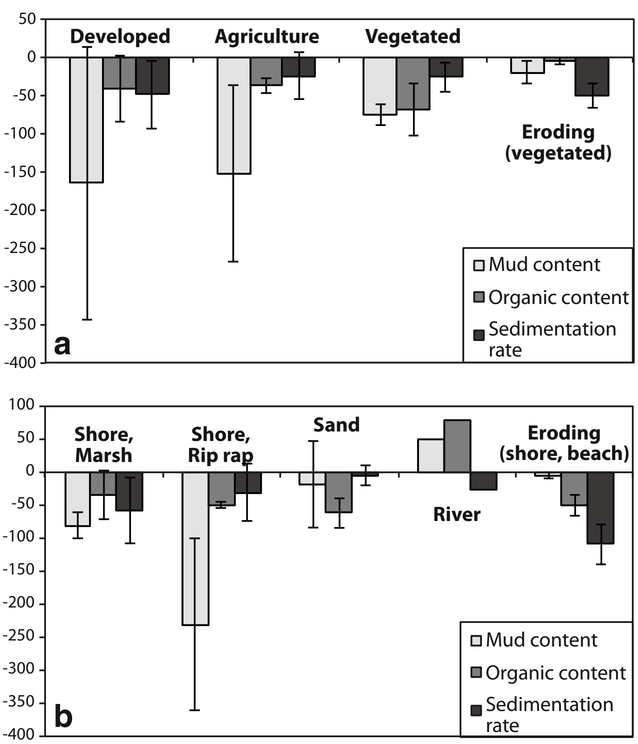

Fig. 6 Average percent change in sedimentary properties for sites grouped by **a** land use and **b** dominant sediment source. In both figures, the rightmost group contains only the naturally eroding shorelines, whereas the other groups contain only sites with structures (i.e., breakwaters, riprap, living shorelines combined). Error bars indicate 1 standard deviation; standard deviation was not calculated for the "river" group, which contains only one site (Elk Neck)

competing model that explained more variability (p = 0.09, adjusted R^2 = 0.52, all coefficients with $p < 0.1$) and included structure type as a variable. Note that structure age was relatively unimportant in these models, while the shoreline-erosion rate and structure type appeared in all models except the simplest model for sedimentation rates.

Discussion

Structural Influences on the Nearshore Environment

While the specific impacts of shoreline stabilization structures clearly depend on the characteristics of individual sites, there are some generalities revealed by this study. For example, sites with the hardest structures (breakwaters) tend to be in higher wave energy, longer fetch environments, while those with softer (living shorelines) alternatives tend to be in more quiescent environments (Fig. 5). This is not surprising, given that harder structures are likely to withstand more energetic conditions than softer alternatives (National Research Council 2007). The breakwater sites in this study had similar erosion rates to the naturally eroding sites, and so the eroding sites effectively represent a control group for high-energy conditions. The primary difference between the presence of breakwaters and uncontrolled shoreline erosion was land use; breakwater sites were either developed or agricultural, while naturally eroding sites were all vegetated (forested or shrubby). This presumably reflects a trade-off between the expense of shoreline stabilization and land value. Wave energy at riprap sites was more like living shoreline sites than breakwater sites. This may imply that living shorelines are a viable alternative to riprap, but not to breakwaters, when shoreline protection is required.

Sedimentary changes at the naturally eroding sites likely reflect broad environmental trends, whereas sedimentary changes at the protected sites also include the influence of structures. In fact, the eroding sites had minimal changes in sediment character over time, except at Elms (note that Cheston Point was consistently erosional over the last ~ 100 years), but sediment mud and organic content decreased for all the riprapped sites and most of the breakwater sites. This difference implicates the structures as agents of change, rather than broader environmental changes. Mud and organic content increased after installation of the breakwater at Elk Neck, which was the only site influenced by a riverine sediment source—the Susquehanna River, which is the major supplier of fine sediment to Chesapeake Bay (Hobbs et al. 1992). Breakwaters are effective sediment traps (Birben et al. 2007; Dolphin et al. 2012; Palinkas et al. 2016), and thus, post-installation changes in sediment character are most related to characteristics of the dominant sediment source. In contrast, riprap installation tends to sever the land-water connection (Runyan and Griggs 2003; Toft et al. 2013). Riprapped sites had developed and

Table 4 Multiple linear regression model statistics for changes in A) mud content, B) organic content, and C) sedimentation rates. For each individual variable listed in the left column of each table, subsequent columns show the regression coefficient, standard error, T-statistic, and *p* value ($p < 0.1$ considered statistically significant). Regression coefficients represent the mean change in the response variable for one unit of change in the predictor variable, while holding other predictors in the model constant. As such, they represent the slope of the linear fit between response and predictor variables, with the sign indicating a positive or negative relationship

Variable	Coefficient	Standard error	T-statistic	*p* value
A) Change in mud content: no models were statistically significant; parameters listed for model with lowest *p* value ($p = 0.32$, adjusted $R^2 = 0.39$)				
Intercept	53.51	19.35	2.77	0.07
Structure age	−0.36	0.24	−1.50	0.23
Structure type – living shoreline	−22.50	6.86	−3.28	0.05
Structure type – riprap	−10.26	4.08	−2.52	0.09
Sediment source – marsh	−5.10	3.28	−1.55	0.22
Sediment source – riprap	−10.09	4.32	−2.34	0.10
Shoreline-erosion rate	34.32	11.49	2.99	0.06
Land use – developed	−9.69	4.49	−2.16	0.12
Land use – vegetated	−3.67	3.47	−1.06	0.37
Parameter	Coefficient	Standard error	T-value	Pr(>|t|)
B) Change in organic content ($p = 0.04$, adjusted $R^2 = 0.63$)				
Intercept	4.99	1.25	3.98	0.007
Structure type – living shoreline	−3.38	0.73	−4.62	0.004
Structure type – riprap	−2.08	0.47	−4.43	0.004
Shoreline-erosion rate	3.85	0.96	4.02	0.007
Land use -developed	−0.75	0.31	−2.47	0.04
Land use - vegetated	−0.46	0.40	−1.15	0.29
C) Change in sediment accumulation rate: parameters for the model with the fewest variables (sediment source and shoreline-erosion rate ($p = 0.09$, adjusted $R^2 = 0.38$) are listed first; parameters for a competing model that explained more variability (also includes structure type; $p = 0.09$, adjusted $R^2 = 0.52$) are listed second.				
Intercept	2.80/4.37	1.07/1.76	2.63/2.49	0.03/0.05
Structure type – living shoreline	−2.04/−0.24	0.67/0.96	−3.02/−0.26	0.02/0.81
Structure type – riprap	−1.25/−1.55	0.66/0.79	−1.89/−1.95	0.9/0.09
Sediment source – marsh	2.49/−2.89	0.88/0.73	2.83/−3.94	0.02/0.008
Sediment source – riprap	NA/−0.57	NA/0.71	NA/0.81	NA/0.45
Shoreline-erosion rate	NA/3.41	NA/1.37	NA/2.48	NA/0.05

agricultural land uses, both of which enhance erosion of terrestrial fine and organic material. These results suggest that delivery of this fine sediment decreased post-installation, leading to decreased deposition in the nearshore.

Mud and organic content also decreased for all sites with living shorelines, except at Aspen, but trends in sedimentation rates were more variable. Sedimentation rates increased for hybrid living shorelines at Aspen and Camp Letts, decreased for the hybrid and soft living shorelines at Patterson Park and CBEC, respectively, and were unchanged for the soft living shorelines at Horn Point and Wye. Increased and unchanged sedimentation rates for hybrid and soft living shorelines, respectively, is intriguing and raises the question of whether differences in structural techniques might be involved. Further examination of this question is hindered by the limited number of sites and serves as an important area for future research, especially since living shorelines are gaining traction as a preferred management alternative (Currin et al. 2010; Shepard et al. 2011).

The regression-model analyses showed that sedimentary changes in the nearshore can be predicted by structure type, shoreline-erosion rate, dominant sediment source, and land use. These parameters were specifically chosen because of their relative ease of determination, although other parameters are also likely insightful. The impact of sedimentary changes to benthic organisms is more difficult to predict, and whether changes are positive or negative depend on specific habitat requirements of individual organisms and/or management goals. This study focused on SAV and found that riprap or breakwater installation had either positive ($n = 3$) or no obvious ($n = 3$) impact on SAV, but negative impacts were observed at one riprapped site; no obvious impacts on SAV were observed at living shoreline sites. In particular, SAV cover increased after structure installation at two sites with sand emplacement (Hoopers Island and St. Marys). This change likely facilitated SAV colonization, since SAV generally prefer coarser substrates (Palinkas and Koch 2012; Swerida 2013). SAV cover also increased at Elk Neck, presumably due to energy reduction landward of the structure. However, note that while the substrate at Elk Neck appears suitable for SAV, continued accumulation of fine and organic material may ultimately prove detrimental. Increased accumulation is hypothesized to be the cause of decreased SAV cover at Severn.

Management Implications

In general, the supply of fine sediment across the land-water boundary appears to decrease when eroding shorelines are protected, at least for riprapped and living shorelines. Most sites were adjacent to marshy shorelines and/or developed or agricultural land-use practices, both of which can supply fine sediment to the aquatic environment. Stabilizing these types of shorelines may thus benefit water quality by decreasing terrestrial fine-sediment loads. However, fine-sediment reduction is not always beneficial; e.g., fine sediments nourish marshes, including fringe marshes in living shorelines, providing a mechanism of resilience against sea-level rise (Davis et al. 2015; Orr et al. 2003; Schile et al. 2014). And, there are other ecosystem costs to shoreline stabilization, especially with hard structures. For example, negative impacts of bulkheads have been well documented and largely linked to wave reflection and corresponding scour of the benthic environment (Bozek and Burdick 2005; Patrick et al. 2016). The impact of riprap on nearshore waves is less clear; a recent study in Chesapeake Bay found that wave energy near riprapped and adjacent natural shorelines was indistinguishable (Sanford unpublished data). Another recent study in Chesapeake Bay showed the relationship of SAV cover to riprap varied by salinity zone. Relative to natural shorelines, SAV occupied similar potential habitat area near riprap in the oligohaline and mesohaline, but less area in the polyhaline (Patrick et al. 2016). The infrequency of adverse riprap impacts in the present study is consistent with these findings, since all the present sites were in the oligohaline or mesohaline Bay. The fact that riprapped shorelines and living shorelines were found in similar wave energy and land use locations may indicate that living shorelines are a viable alternative to riprap where shoreline protection is required and wave energy is moderate. Note that the present study did not include sandy shorelines, and so the water-quality impact of stabilizing these types is unclear. However, we did observe that nearshore sand emplacement benefitted SAV, which generally prefer coarser substrates, implying the erosion of sandy shorelines may also be beneficial to SAV.

While this broad comparative study was built largely on existing data and is thus limited in its conclusions, it provides much valuable insight for future research. For example, future work might focus on more detailed comparisons of fewer shoreline protection alternatives; e.g., are living shorelines truly viable alternatives for riprap, and if so, under what conditions? Future work might also focus on impacts from single protection techniques; e.g., how do living shorelines impact SAV? The current study serves as an important advance towards addressing these questions, which is critical for effective shoreline management.

Conclusion

The main conclusion of this study is that there is no "one size fits all" answer to the question of anticipated changes to the nearshore sedimentary environment associated with shoreline stabilization structures. None of the general trends in changes in mud/organic content or sedimentation rates were statistically significant, and no consistent relationships were observed between these changes and structure age. However, guidance on expected changes at individual sites can be provided, given such site characteristics as structure type, historical shoreline-erosion rate, land use, and dominant sediment source. For example, sediment mud and organic content decreased for all the riprapped sites and most of the breakwater sites. Breakwaters serve as effective sediment traps, and so post-installation changes in sediment character are driven by characteristics of the dominant sediment source. In contrast, riprap installation tends to sever the land-water connection, reducing delivery of terrestrial sediment to the nearshore. The land uses adjacent to the riprapped sites (development and agriculture) tend to enhance delivery of fine and organic material across the land-water interface, likely from decreased supply after riprap installation. Mud and organic content also decreased for most sites with living shorelines, but trends in sedimentation rates were equivocal and may be related to differences in structural techniques. Riprap or breakwater installation had either positive (n = 3) or no obvious (n = 3) impact on SAV at most sites (total of seven sites) but negative impacts were observed at one riprapped site; no obvious impacts on SAV were observed at living shoreline sites. In particular, SAV cover increased after structure installation at sites with sand emplacement, but SAV cover decreased at a site with increased accumulation after structure installation. Finally, there were clear associations between structure installation, land use, and wave exposure, as might be expected. Almost all (85%) the protected shoreline sites were developed or agricultural, while all the eroding shorelines were vegetated. Similarly, eroding and breakwater sites experienced the highest wave exposure, while riprap and living shoreline sites were less exposed.

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