

# Ecological tradeoffs of stabilized salt marshes as a shoreline protection strategy: Effects of artificial structures on macrobenthic assemblages



D.M. Bilkovic\*, M.M. Mitchell

Virginia Institute of Marine Science, College of William & Mary, P.O. Box 1346, Gloucester Point, VA 23062, USA

## ARTICLE INFO

### Article history:

Received 18 June 2013

Received in revised form

26 September 2013

Accepted 12 October 2013

### Keywords:

Benthic communities  
Chesapeake Bay  
Created wetlands  
Erosion control  
Estuarine habitats  
Fringing marsh  
Living shoreline  
Marsh-sill  
Restoration  
Shallow-subtidal  
Shoreline hardening

## ABSTRACT

Armoring shorelines to prevent erosion is a long-standing global practice that has well-documented adverse effects on coastal habitats and organisms. A relatively new form of shoreline protection, referred to as hybrid stabilization, incorporates created marsh in combination with a stabilizing structure such as a low-profile stone sill and is being implemented in many US coastal states as a means to not only control erosion but also to restore coastal habitat. However, there has been limited scientific investigation of ecological benefits and impacts associated with implementation of hybrid stabilization. We evaluated relative habitat capacity of marsh-sills by comparing plant, sediment, and benthic macroinvertebrate attributes in intertidal and subtidal zones of existing marsh-sills, natural marshes, tidal flats, and riprap revetment within two subestuaries of Chesapeake Bay, USA. Low and high marsh plant characteristics (stem count and height) of marsh-sills were similar to or greater than natural marshes. However, sediment was coarser, total organic carbon and total nitrogen concentrations were lower, and benthic macrofaunal community structure differed in marsh-sills compared to natural marshes. Marsh-sills supported lower deposit-feeding infaunal biomass than marshes in the intertidal. Epifaunal suspension-feeders were most prevalent at sites with artificial structure (riprap and marsh-sill), but highly variable among subestuaries. Infaunal abundance, biomass, diversity, and proportion of suspension/interface and deposit feeding animals were greater in shallow subtidal than in intertidal environments. Conversion of existing habitat to marsh-sills may cause localized loss of benthic productivity and sediment bioturbation and nutrient-cycling functions, with the opportunity to enhance filtration capacity by epifaunal recruitment to structures. When creating marshes that require structural support, there should be a balance of minimizing loss of existing habitats while encouraging use of suitable structural habitat for suspension-feeders. If properly implemented, the addition of structural habitat could subsidize secondary productivity particularly in areas where loss of complex biogenic habitat (e.g., oyster reefs) has occurred.

© 2013 Elsevier B.V. All rights reserved.

## 1. Introduction

As human populations increase, the balance between preserving ecosystem functions while allowing for the creation and protection of human infrastructure has become an increasing challenge. With ~10% of the world's populations living in the low-elevation coastal zone (McGranahan et al., 2007), there is intense pressure to protect human infrastructure from environmental pressures such as wind- and tide-driven shoreline erosion, resulting in a global practice of hardening shorelines, typically using concrete and wood bulkheads or rock revetments. While these structures can slow shoreline erosion, they are still vulnerable to storm damage and, increasingly, to sea level rise. In addition, they act as an ecological

barrier, separating upland and wetland functions, increasing the rate of wetland loss, and changing the character of the nearshore ecosystem (Peterson and Lowe, 2009).

Armoring shorelines can result in fragmentation or loss of habitats (Peterson and Lowe, 2009; Dugan et al., 2011), reduction in capacity to moderate pollutant loads delivered to coastal waters (NRC, 2007), increased scouring and turbidity (Bozek and Burdick, 2005), reduction in nekton and macrobenthic invertebrate diversity and community integrity (Peterson et al., 2000; Chapman, 2003; King et al., 2005; Bilkovic et al., 2006; Seitz et al., 2006; Bilkovic and Roggero, 2008; Morley et al., 2012), increases in invasive species, such as *Phragmites australis* (Chambers et al., 1999; King et al., 2007), and disturbance of sediment budgets sustaining adjacent properties (Bozek and Burdick, 2005; NRC, 2007). This has led to an interest in marsh restoration and creation, frequently paired with a reduced form of hardened structure, as a method for lessening shoreline erosion while maintaining natural ecosystem

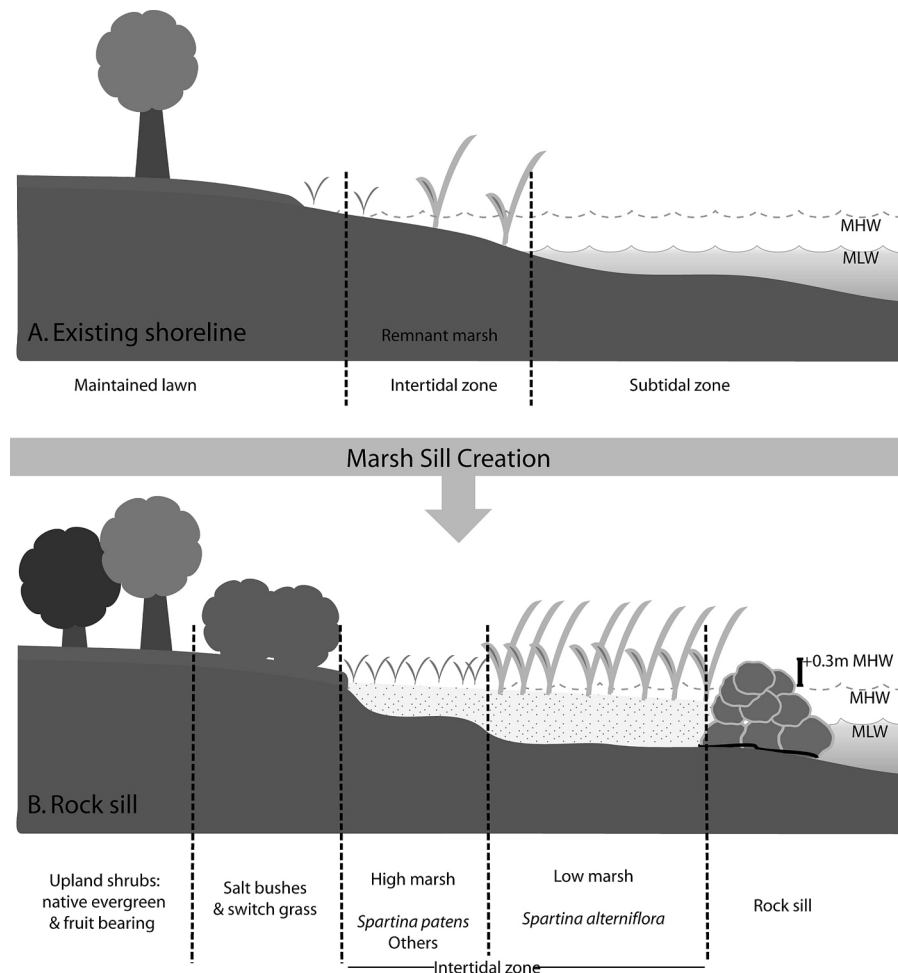
\* Corresponding author. Tel.: +1 804 684 7331; fax: +1 804 684 7179.  
E-mail address: [donnab@vims.edu](mailto:donnab@vims.edu) (D.M. Bilkovic).

function. Many US states including Virginia, Maryland, North Carolina, Texas, Alabama and Florida have enacted regulations to encourage or require the use of “living shoreline” approaches (non-structural or hybrid stabilization) instead of bulkheads or seawalls (CCRM, 2010; Currin et al., 2010). However, there has been limited scientific investigation of ecological benefits and impacts associated with the implementation of natural shoreline protection designs (Burke et al., 2005; Davis et al., 2006; Currin et al., 2010). This is particularly true for hybrid stabilization projects in higher energy systems that include rock structure with a planted marsh, such as marsh-sills (low, free standing stone structures placed parallel to and near the marsh shoreline). Uncertainty remains about the ecological tradeoffs involved in the conversion of existing habitat to a marsh-sill.

Hybrid stabilization approaches rely on the extensive body of literature on marsh creation, which provides evidence that created marshes attain comparable functions to natural marshes over time. In created marshes, most ecological attributes reportedly follow a predictable trajectory toward structural/functional equivalence to natural marshes. Within 5–15 years, primary producers and macrobenthic infaunal communities typically reach equivalence, while organic carbon and nitrogen accumulation may require in excess of 25 years (Craft et al., 1999, 2003). The untested assumption is that marshes associated with artificial rock structures (sills) will act as created marshes and the placement of sills is ecologically neutral

or beneficial. Unfortunately, research on created marshes primarily has focused on extensive meadow marshes instead of the narrow fringing marshes along a shoreline which typify a marsh-sill (Currin et al., 2008). These fringing marshes are typically less than 20 m wide and are comprised of narrow bands of low marsh *Spartina alterniflora* and high marsh *Spartina patens* vegetation. Narrow fringing and extensive marshes provide some equivalent functions. For example, wave reduction and sediment deposition may occur within a short distance in a marsh (<10 m), signifying that fringe marshes may be effective at wave dampening (Morgan et al., 2009) and sediment retention (Christiansen et al., 2000; Neubauer et al., 2002; Morse et al., 2004) if the amount of edge habitat is similar to extensive marshes. For other functions, such as secondary production, uncertainty remains as to whether fringing marshes provide comparable habitat as extensive marshes (Morgan et al., 2009). Further, few studies have evaluated the ecological attributes (e.g., faunal community structure, sediment composition, plant characteristics) of created marshes with sills in relation to natural marshes (Currin et al., 2008, 2010) and to the best of our knowledge only one has evaluated benthic infaunal communities (Wong et al., 2011).

Typical marsh-sill construction involves a combination of bank grading and filling of intertidal and shallow water subtidal area to create appropriate elevations for marsh vegetation. Rock structures, placed at the channelward edge of the created marsh, are often used in higher energy environments where some protection



**Fig. 1.** Cross-section of characteristic placement of a marsh-sill on a shoreline. A marsh-sill is often built channelward with conversion of existing unvegetated wetland (flats) and subtidal lands to sand fill, planted marsh, and sill structure. These designs result in a wider intertidal area and a change in elevation, sediment type, and plant usage but better maintain the upland-water connection compared to riprap revetments and bulkheads. Mean high water (MHW), mean low water (MLW).

is needed to support plant growth. In regions where oyster shell is readily available and oysters colonize intertidal elevations (e.g., North Carolina), bagged oyster shell may be used instead of rock structure. The supporting structure may be continuous or in sections with tidal openings (henceforth referred to as “gaps”) to allow nekton access to the marsh. The placement of a marsh-sill involves the conversion of existing intertidal and shallow subtidal habitat into an intertidal vegetated fringing marsh with a rock structure (Fig. 1). In Virginia from 2009 to 2011, the types of shorelines that were most frequently converted to marsh-sills ( $n=81$ ) were eroding relict fringe marshes and/or unvegetated tidal flats (98%) and to a lesser degree failed or failing bulkheads or riprap revetment (2%) (CCRM, 2012). In addition, 2.5 ha (6.2 acres) of shallow subtidal bottom were converted to intertidal marsh-sill habitat. This area represents a relatively small proportion of the total shallow water ( $\leq 2$  m depth) in the meso-polyhaline regions of Virginia ( $\sim 0.003\%$  of 92,280 ha); however, this number may be misleading because in some locations shallow water areas are limited and impacts may be proportionally greater (Bilkovic et al., 2009).

The potential benefit of constructing a marsh-sill and the ecological tradeoffs therein will depend in part on what existing habitat is being replaced. Those existing shallow habitats adjacent to unaltered shorelines often support highly productive benthic microalgal communities that contribute significantly to primary production in estuaries (MacIntyre et al., 1996; Miller et al., 1996), are important to nutrient cycling (Tyler et al., 2003), support higher trophic levels (Middelburg et al., 2000), and maintain sediment stability (Madsen et al., 1993; Underwood and Paterson, 1993). The unvegetated intertidal and shallow subtidal areas also provide refuge and feeding habitat for juvenile fish and invertebrates (Ruiz et al., 1993). Armored shorelines are characterized by a reduction in shallow-water refuge area and have been linked to a loss of diversity and reduction of benthic and nekton community structure and integrity (Bilkovic et al., 2006; Seitz et al., 2006; Bilkovic

and Roggero, 2008). This suggests that the net benefit of constructing a marsh-sill may be greater in cases where the alternative is construction of an armored shoreline than in cases where the marsh-sill replaces existing natural habitats.

The purpose of this study was to assess relative change in habitat capacity (the biological community structure and diversity supported by the habitat), which occurs as a result of marsh-sill implementation for upland erosion protection. We specifically addressed the following questions: (1) Are the ecological attributes (i.e., sediment, marsh plant, and benthic macrofaunal community characteristics) of marsh-sills similar to those of natural shorelines? (2) What are the ecological tradeoffs (shifts in benthic macrofaunal communities) of converting existing intertidal and shallow subtidal habitat to vegetated marsh-sill habitat? (3) Can marsh-sills be an effective form of coastal habitat restoration?

## 2. Methods

### 2.1. Study area and sampling design

Our study areas were located in two meso-polyhaline subestuaries of Chesapeake Bay: East River (Mathews County, VA), and South River (Anne Arundel County, MD) (Fig. 2). In Mathews County, 11% (60 km) of the tidal shoreline is hardened, with 37% (202 km) of riparian lands developed (Berman et al., 2000, 2006). In Anne Arundel County, 43% (327 km) of the tidal shoreline is hardened, with 64% (487 km) of riparian lands developed. In each subestuary (river), we conducted a paired-site comparison of a marsh-sill in relation to natural and hardened shoreline types (natural fringing marsh, unvegetated tidal flat, and riprap revetment). To select marsh-sill sites, we initially inventoried all marsh-sill projects in Chesapeake Bay between 2 and 10 years of age (to ensure at least 2 growing seasons and modern construction standards)

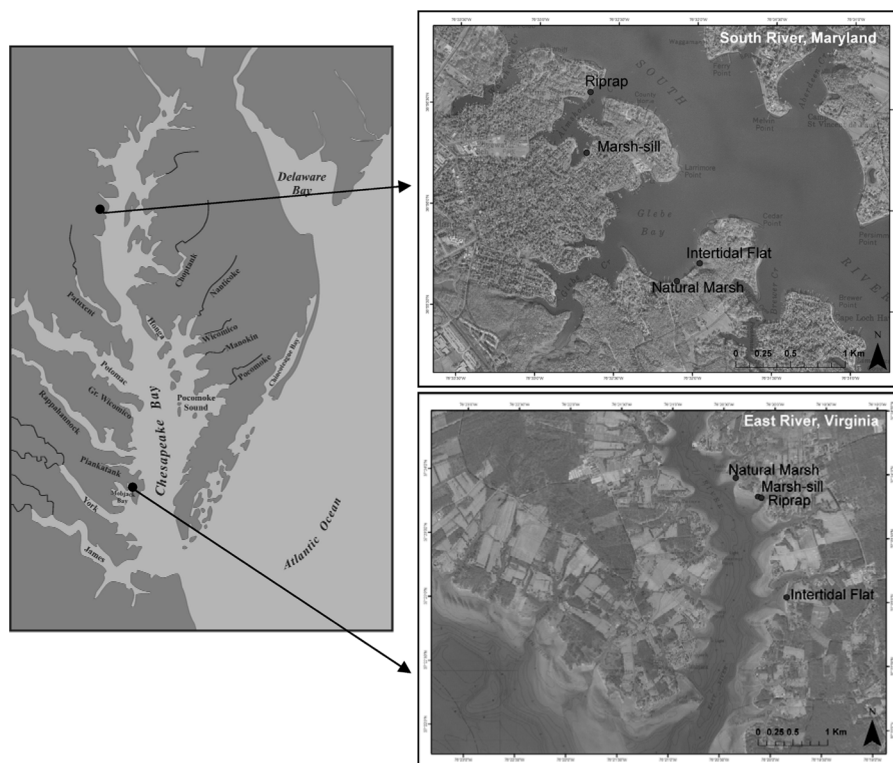


Fig. 2. Map of study sites in Chesapeake Bay. Locations of shoreline types (marsh-sill, natural marsh, intertidal flat, and riprap revetment) in the East River, Virginia and the South River, Maryland are depicted in map insets.

**Table 1**

Physical characteristics of shoreline sites. Resid = residential land use. Units in meters unless noted otherwise.

|                       | East River              |          |          |            | South River                            |          |       |            |
|-----------------------|-------------------------|----------|----------|------------|----------------------------------------|----------|-------|------------|
|                       | Marsh-sill              | Marsh    | Flat     | Riprap     | Marsh-sill                             | Marsh    | Flat  | Riprap     |
| Site length           | 256                     | 73       | 61       | 91         | 244                                    | 61       | 30    | 70         |
| Riparian land use     | Resid                   | Resid    | Resid    | Resid      | Resid & Road                           | Forested | Resid | Resid      |
| Wave energy           | Moderate                | Moderate | Moderate | Moderate   | Low                                    | Moderate | Low   | Moderate   |
| Widest fetch (NM)     | 1–5                     | 1–5      | 1.2      | 1–5        | <1                                     | 1.2      | <1    | 1.3        |
| Orientation           | SW                      | SW       | W        | SW         | NW                                     | NW       | NW    | SE         |
| Avg slope %           | 1.3                     | 2.0      | 2.7      | 6.9        | 5.6                                    | 3.4      | 1.9   | 9.9        |
| Structure length      | 256                     | –        | –        | 91         | 122                                    | –        | –     | 70         |
| Build date            | 2003–2004               | –        | –        | 2003–2004  | 2008                                   | –        | –     | ~2009      |
| Structure description | 3 sills with 2.5 m gaps | –        | –        | Continuous | 5 sills & equidistant sill/gap pattern | –        | –     | Continuous |
| Marsh length          | 256                     | 73       | –        | –          | 122                                    | 61       | –     | –          |
| Ave low marsh width   | 3.8                     | 3.2      | –        | –          | 6.1                                    | 1.3      | –     | –          |
| Ave high marsh width  | 6.7                     | 18.3     | –        | –          | 4.3                                    | 13.8     | –     | –          |

using Maryland Department of the Environment, Maryland Department of Natural Resources, and Virginia Institute of Marine Science (VIMS) shoreline permit databases. Besides the age of the project, sites were required to (1) be within the meso-polyhaline salinity regime, (2) possess neighboring (within the same subestuary) shorelines with natural *Spartina*-dominated fringe (<20 m wide) salt marsh, unvegetated flats, and riprap revetment for comparison, (3) be accessible, and (4) meet certain design standards (planted marshes protected by a sill built of rocks not exceeding 300 lbs with a crest  $\leq 0.3$  m above mean high water (MHW) and regular gaps to allow tidal intrusion). On that basis, the candidate pool was narrowed and permission was obtained for sampling at two marsh-sill sites – one in the South River, MD that was built in 2008 and one in the East River, VA that was built in 2003–2004. At the time of sampling, the South River marsh-sill was between 2 and 3 years old and the East River marsh-sill was between 7 and 8 years old.

We selected neighboring comparative sites (marsh, flats, riprap) from a pool of candidate sites in each subestuary that were identified using remotely sensed data (digital shoreline inventories and aerial imagery). Sites were required to meet several criteria to reduce confounding factors known to influence benthic species distribution, including comparable salinity and energy regimes, minimum length of shoreline ( $\geq 30$  m contiguous shoreline condition), sediment characteristics (predominantly sand), and accessibility (Table 1). There were limited natural habitats (fringing marsh, flats) within both subestuaries that met the criteria, so selected sites were representative of remaining natural habitats. Sampling was carried out in September 2010 and May–June 2011 to capture seasonal variability in macrofaunal populations that are influenced by shifts in predation and food availability throughout the year (Holland et al., 1977; Hines et al., 1990).

For each site ( $n=8$ ), we randomly selected transects that ran perpendicularly from the shore to the shallow subtidal zone (between 0.6 and 1 m relative to mean low water) and were at least 2 m apart. In each river, we equally sampled five distinct shoreline habitats for comparisons: natural marsh, intertidal flat, sill present, tidal gap, and riprap revetment (Fig. 3). The marsh-sills contain low-profile, shore-parallel, offshore rock structures placed in sections with tidal gaps and are designed to protect newly planted wetland vegetation. For benthic macrofaunal community analyses, we considered marsh-sills to consist of two distinct shoreline habitats (sill present and tidal gaps) because of differing tidal flow, potentially affecting recruitment opportunities. We defined ‘sill present’ as habitat where the rock sill runs parallel to the shoreline protecting the marsh and ‘tidal gaps’ as habitat where there is an opening in the rock sill. Riprap revetments were sloped structures, consisting of multiple layers of stone constructed against a bank,

which cover the intertidal. Natural marshes consisted of *S. alterniflora* dominated fringing (<20 m wide) marsh habitats. Intertidal flats were habitats of unvegetated sandy bottom between MHW and mean high water (MLW). We sampled 6 transects within each shoreline habitat in September 2010 and 9 transects in May–June 2011. For each transect, hydrographic measures and macrobenthic infaunal communities were sampled once in the intertidal and once in the subtidal zone. Intertidal infaunal and physicochemical samples were obtained from the seaward edge of the low marsh at the marsh-sill and marsh sites and within 5–10 cm water depth at mean low water at the intertidal flats. Intertidal habitat bottom was unavailable at riprap revetment sites because the riprap footprint covers the intertidal. Subtidal samples were obtained in depths between 0.6 and 1 m relative to mean low water. Plant characteristics were assessed in low (*S. alterniflora*) and high (*S. patens*) marsh zones. Epifauna were sampled on shoreline stabilization structures (sill, riprap) and wetland plants and surfaces (edge of low marsh or tidal flats).

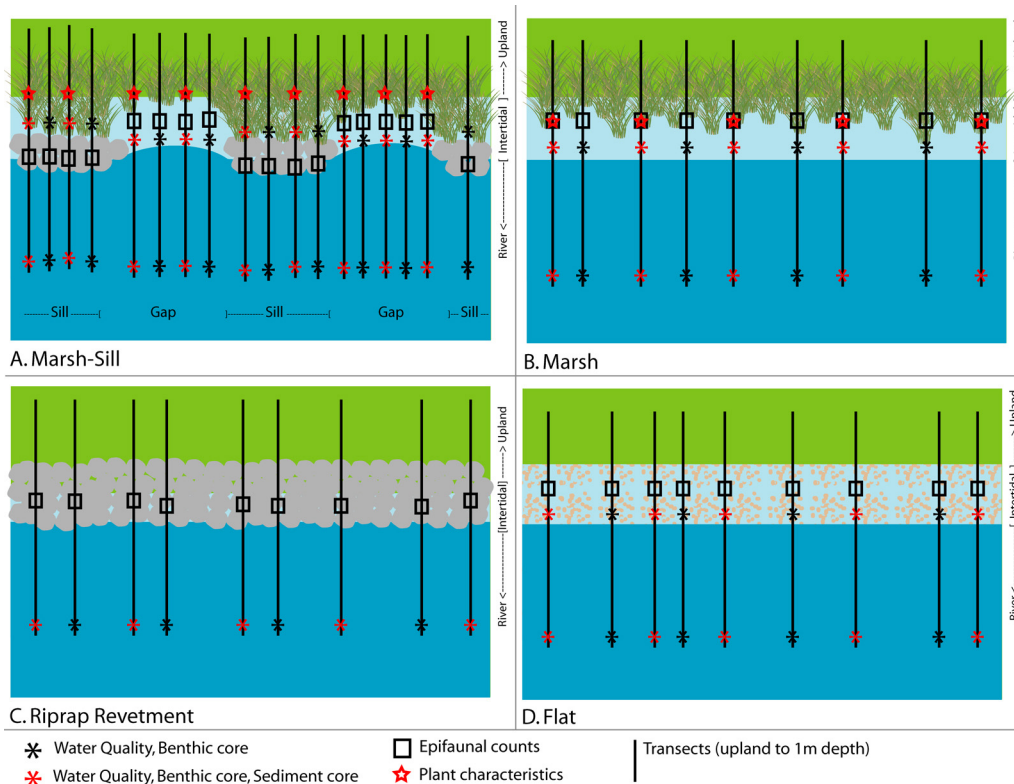
## 2.2. Data collection

### 2.2.1. Physical attributes, hydrographic conditions, and plant characteristics

During each infauna sampling event we measured physical and hydrographic variables that may influence benthic faunal distribution and abundance. We used a hand-held YSI sonde to record dissolved oxygen (DO), salinity, conductivity, pH, turbidity, water temperature, and chlorophyll *a* ~0.3 m above the bottom. For each shoreline habitat, we collected a sediment core near 6 infauna sample locations (three cores in the intertidal and three cores in the subtidal) and determined the percentages of grain-size, total organic carbon (TOC), total nitrogen (TN), and organic matter (OM) content within intertidal and subtidal zones. Percentages of gravel, sand, silt, and clay in sediments were determined by standard wet sieve and pipette analysis (Folk, 1980) and C and N quantified using an Exeter CE440 elemental analyzer at the Virginia Institute of Marine Science (VIMS) Analytical Services Center.

We evaluated geomorphic characteristics onsite and remotely: slope, relative wave energy, fetch, orientation, structure length, and riparian land use. We calculated the slope (distance from the upland border to water depths of 0.6–1.0 MLW) at 3 transects per habitat. We determined fetch, shoreline orientation, and structure (i.e., sill or riprap) length from aerial photography in ArcGIS, and assessed riparian land use and wave energy on-site. In low (*S. alterniflora*) and high (*S. patens*) marsh zones in 2011, we identified and enumerated plants and measured plant height of the 3 tallest stems within 0.25 m<sup>2</sup> quadrats placed randomly at 3 transects





**Fig. 3.** Schematic representation of the sampling design. Five habitat types were compared in each subestuary: natural fringing salt marsh, tidal flats, marsh-sill with sill present, marsh-sill tidal gaps, and riprap revetment. For each subestuary, 6 transects in 2010 and 9 transects in 2011 were sampled for each of the 5 habitats. Figure illustrates 2011 sampling.

per habitat. We measured the area of tidal inundation (marsh width) at 3 transects per habitat, and estimated width using aerial photography for larger marsh extents.

### 2.2.2. Benthic macrofaunal community

For each transect we took one core sample (30 cm depth, 10 cm diameter) from the intertidal and one from the subtidal zone and sieved sediment on a 0.5-mm mesh. We sorted all samples and identified infauna down to the lowest practical taxonomic unit (generally species). We dried specimens to a constant weight (typically for 48 h) at 60 °C and ashed at 550 °C for 4 h to obtain ash free dry weight (AFDW). We ashed each taxon separately and shucked large bivalves prior to ashing to remove additional weight of the periostracum.

We sampled epifauna within the intertidal zone of each site concurrently with infaunal sampling. For each transect, we identified and enumerated each epifauna species present on the shoreline stabilization structures (seaward side) within a 0.25-m<sup>2</sup> quadrat, if present, or on the wetland surface when structures were absent. We recorded an image of each quadrat for visual analyses in the lab. Because epifauna were not removed in order to reduce site disturbance, we estimated species-specific biomass (AFDW, mg m<sup>-2</sup>) by applying an estimated average size to published L-W relationships for each enumerated species. We used field observations and visual inspection of quadrat images to roughly estimate average sizes and verify that the size structure for each observed epifaunal species was similar among sites compared. For eastern oysters *Crassostrea virginica*, we applied a shell length value of 80.5 mm (age 2) representing the average size of oysters observed (East River marsh, marsh-sill, and riprap sites) to an oyster L-SFDW (shell-free dry weight) equation from Mann et al. (2009) and estimated AFDW from SFDW using a conversion factor of 83.5% (Ricciardi

and Bourget, 1998). For ribbed mussels *Geukensia demissa*, we applied an average size of 8 cm, representing the average size of mussels observed (East River marsh and marsh-sill sites), to a L-SFDW equation (Hughes and Seed, 1981) and estimated AFDW with the above conversion factor. For hooked mussels *Ischadium recurvum*, we applied an average size of 36 mm to a L-AFDW regression (Berlin, 2008) representing the average size of hooked mussels observed (East River marsh-sill and riprap sites). For barnacles we recorded the number of animals in the field and determined the mean AFDW in the laboratory for two size classes (small: 3–9 mm and medium-large: 10–25 mm) and used these values to estimate biomass.

### 2.3. Data analysis

#### 2.3.1. Physical attributes, hydrographic conditions, and plant characteristics

Because of anticipated regional and tidal zone differences in species compositions, we analyzed data from each river (East, South) and tidal zone (intertidal, subtidal) separately. We compared physical habitat attributes across shorelines within each river using 1-way ANOVAs with shoreline (marsh, flat, marsh-sill, riprap) as a fixed factor and grain size, percent organic matter, percent TOC and TN of sediments, or chlorophyll *a* as the dependent variable. We compared low (*S. alterniflora*) and high (*S. patens*) marsh plant characteristics (stem height, density) between natural marshes and marsh-sill using a 1-way fixed factor ANOVA. We tested normality of response variables using the Kolmogorov–Smirnov test and homogeneity of variances using Levene's test. When necessary, data were log transformed and retested. When ANOVA results showed differences, we used a

Tukey's HSD post hoc test for multiple comparisons. Statistical analyses were done using SPSS v.20 software.

### 2.3.2. Benthic infaunal and epifaunal communities

Benthic community data were analyzed using permutational analysis of variance (PERMANOVA), a semi-parametric procedure for analyzing univariate or multivariate data (Anderson et al., 2008). Henceforth, univariate PERMANOVA refers to comparisons on aggregate data such as total abundance and multivariate PERMANOVA refers to comparisons of community structure.

We estimated total and species-specific abundance (number of individuals  $m^{-2}$ ) and biomass (AFDW,  $mg\ m^{-2}$ ) of benthic infauna and epifauna separately for each site, shoreline habitat, and tidal zone. We placed macrofauna into feeding guilds (suspension, interface, deposit, carnivore/omnivore) following Dauer et al., 2002. To determine the effects of shoreline habitat (marsh, flat, sill, gap, riprap) and year (2010, 2011) on macrofaunal invertebrate communities, we used multivariate and univariate PERMANOVA. We tested differences in the total infaunal abundance, biomass, species diversity (taxonomic distinctness, AvTD), the total biomass of infaunal suspension/interface and deposit feeders, and epifaunal (intertidal only) abundance and biomass with univariate PERMANOVA. Taxonomic distinctness is a measure of taxonomic relatedness of the species in each sample and has several advantages over traditional diversity measures such as species richness including insensitivity to sampling effort, ability to estimate phylogenetic diversity, incorporation of evenness measures, and being more closely linked to functional diversity (Clarke and Warwick, 1999).

To assess habitat conversion tradeoffs for macrofaunal communities, we compared biomass (AFDW) of infaunal suspension/interface and deposit feeders in areas converted to marsh-sill intertidal (sill and gap habitats combined) with those areas that have the potential to be converted to marsh-sill intertidal: (1) intertidal habitat of marsh, flat, and riprap shorelines, and (2) shallow subtidal habitat adjacent to marsh, flat, and riprap shorelines. Comparisons were made at the shoreline level (i.e., marsh-sill, marsh, flat, riprap) to evaluate the total effect of conversion. To test the effects of tidal zone on infauna, we compared total abundance and biomass between the intertidal and subtidal with univariate PERMANOVA. We ran univariate PERMANOVA tests on Euclidean distances matrices of log-transformed data using a Type III sum of squares and 4999 permutations. We tested differences in infaunal community structure among shoreline habitat and between years for each river using multivariate PERMANOVA. We ran multivariate PERMANOVA tests on Bray–Curtis similarity matrices of log-transformed species abundance and biomass data using a Type III sum of squares and 4999 permutations (PRIMER-E v6 with PERMANOVA extension, Clarke and Warwick, 1999).

## 3. Results

### 3.1. Physical attributes, hydrographic conditions, and plant characteristics

Riprap revetment sites had the steepest offshore slopes relative to paired sites. The slope of the sill and natural sites varied more in the South River than in the East River which had sites with more similar, gradual slopes (Table 1). Hydrographic parameters (water temperature, dissolved oxygen, and salinity) were comparable between paired marsh-sills and natural wetlands for each year (Table 2). Chlorophyll *a* concentrations were similar between marsh-sills and natural marshes in the intertidal and the East River subtidal (Tukey's HSD > 0.05); but chlorophyll *a* was higher in

the marsh ( $37.4\ \mu g\ L^{-1}$ ) than marsh-sills ( $27.6\ \mu g\ L^{-1}$ ) ( $F_{3,45} = 14.6$ ,  $P < 0.0001$ ) within the South River subtidal (Table 2).

Sediments at all sites were predominantly composed of sand (mean of 84% in intertidal and 91% in subtidal) with organic matter content ranging between 0.2 and 4.8% in the intertidal and 0.4–2.3% in the subtidal. Sediment grain-size in the intertidal was larger in marsh-sills (7.1 and 21.5% gravel) than natural marshes ( $\sim 0.1\%$ ) ( $F_{2,12} = 7.2$  and  $11.2$ , East and South Rivers respectively,  $P < 0.05$ ). Within the East River intertidal organic matter was lower at the marsh-sill than natural marsh ( $F_{2,24} = 14.6$ ,  $P < 0.0001$ ), but did not differ between the sill and natural marshes in the South River ( $F_{2,24} = 2.2$ ,  $P = 0.14$ ). TOC and TN were higher in the intertidal sediment of natural marshes than marsh-sills (TOC:  $F_{2,12} = 76.4$  and  $5.6$ ; TN:  $F_{2,12} = 13.1$  and  $3.6$ ,  $P < 0.05$ ), while subtidal sediments had similar TOC and TN ( $P > 0.05$ ) (Table 2).

In natural and sill-marshes, the predominant species (>98%) were *S. alterniflora* (low marsh) and *S. patens* (high marsh). Low marsh (*S. alterniflora*) mean plant density ranged from 132 to 464 stem  $m^{-2}$  and did not differ significantly between marsh-sills and natural marshes ( $F_{1,9} = 0.4$  and  $4.2$ ;  $P > 0.05$ ). *S. alterniflora* height (cm) was similar between the marsh-sill and natural marsh in the East River ( $F_{1,9} = 0.7$ ,  $P = 0.45$ ), but in the South River marsh-sill plants were taller ( $81.2\ cm \pm 11.1\ SD$ ) than natural marsh plants ( $41.0\ cm \pm 12.8\ SD$ ;  $F_{1,9} = 24.1$ ,  $P = 0.02$ ; Fig. 4). High marsh (*S. patens*) mean plant density (stems  $m^{-2}$ ) was higher in sill marshes (1746 (East), 1275 (South)) than paired natural marshes (755, 644) ( $F_{1,9} = 24.0$  and  $8.0$ ,  $P < 0.05$ ). Similar to low marsh characteristics, high marsh plant height was similar between the marsh-sill and natural marsh in the East River ( $F_{1,9} = 0.4$ ,  $P = 0.56$ ), but in the South River marsh-sill plants ( $72.0\ cm \pm 15.5\ SD$ ) were taller than those in the natural marsh ( $43.7\ cm \pm 10.1\ SD$ ;  $F_{1,9} = 8.0$ ,  $P = 0.03$ ; Fig. 4).

### 3.2. Benthic macrofaunal communities

In 2010, in 223 core samples, we captured 821 infauna belonging to 36 species and 22 families. In 2011, in 162 core samples, we captured 490 infauna belonging to 42 species and 21 families. On average, annelids dominated samples (51.8%), followed by arthropods (18.7%), molluscs (10.4%), and phoronids (4.3%). Phoronids were present only in the East River and mainly in subtidal zones; unusually high numbers occurred along the marsh-sill site (32.5%) as compared to other shorelines (<3%). The number of species captured in each core varied from 0 to 7 ( $1.6 \pm 1.4$  [mean  $\pm$  SD]) and the number of individuals  $m^{-2}$  varied from 0 to 905 ( $38.8 \pm 67.9$ ). Subtidal zones had a higher number of species (40, 16 unique to subtidal) than the intertidal (25, 2 unique to intertidal).

Suspension-feeding oysters, barnacles, and mussels made up  $\geq 98\%$  of the epifauna present. Predominant epifauna at the East River marsh-sill and riprap were eastern oyster (*C. virginica*), hooked mussels (*I. recurvum*) and barnacles (*Balanus* and *Chthamalus* spp.) ( $111.5 \pm 51.0$ ,  $12.9 \pm 11.4$ ,  $351.2 \pm 233.6$  individuals  $m^{-2}$ , respectively); the natural marsh was comprised of eastern oysters and Atlantic ribbed mussels (*G. demissa*) ( $32.0 \pm 38.5$ ,  $20.5 \pm 27.4$  individuals  $m^{-2}$ , respectively). Within the South River, the only epifauna species observed were *Balanus* at the marsh-sill and riprap sites ( $54.3 \pm 66.8$  individuals  $m^{-2}$ ).

### 3.3. Total abundance, biomass, and diversity

Infaunal abundance and biomass were lower within intertidal marsh-sill habitats compared to natural wetlands on the East River, but similar on the South River (Supplementary Table 1 and

**Table 2**

Mean hydrographic conditions and sediment characteristics by river and shoreline habitat.

|                                        | Intertidal |       |      |      |        | Subtidal |      |      |      |        |
|----------------------------------------|------------|-------|------|------|--------|----------|------|------|------|--------|
|                                        | Marsh      | Flat  | Sill | Gap  | Riprap | Marsh    | Flat | Sill | Gap  | Riprap |
| <b>East River</b>                      |            |       |      |      |        |          |      |      |      |        |
| Water temperature (°C)                 |            |       |      |      |        |          |      |      |      |        |
| September 2010                         | 26.2       | 24.4  | 25.4 | 25.6 | –      | 26.0     | 24.9 | 25.5 | 25.7 | 27.0   |
| May/June 2011                          | 28.3       | 26.8  | 26.6 | 26.8 | –      | 28.3     | 28.4 | 29.4 | 29.4 | 28.0   |
| Salinity                               |            |       |      |      |        |          |      |      |      |        |
| September 2010                         | 22.1       | 22.1  | 22.0 | 22.0 | –      | 22.0     | 22.0 | 22.2 | 22.4 | 22.0   |
| May/June 2011                          | 14.7       | 14.7  | 14.7 | 14.6 | –      | 14.6     | 14.2 | 14.3 | 14.4 | 14.8   |
| Turbidity (NTU)                        |            |       |      |      |        |          |      |      |      |        |
| September 2010                         | 18.9       | 19.2  | 11.8 | 7.9  | –      | 19.2     | 13.5 | 12.4 | 9.9  | 24.2   |
| May/June 2011                          | 37.1       | 137.4 | 79.8 | 9.4  | –      | 12.9     | 11.1 | 7.6  | 8.7  | 8.9    |
| Dissolved oxygen (mg L <sup>-1</sup> ) |            |       |      |      |        |          |      |      |      |        |
| September 2010                         | 6.5        | 7.3   | 6.8  | 6.9  | –      | 6.4      | 7.3  | 7.4  | 7.5  | 7.9    |
| May/June 2011                          | 7.2        | 7.5   | 6.9  | 6.4  | –      | 7.2      | 7.2  | 9.2  | 9.1  | 9.2    |
| Sand + gravel (%)                      | 81.8       | 88.0  | 94.4 | 89.0 | –      | 93.5     | 91.7 | 89.2 | 79.8 | 90.2   |
| Silt + clay (%)                        | 18.2       | 12.0  | 5.6  | 11.0 | –      | 6.5      | 8.3  | 10.8 | 20.3 | 9.8    |
| Organic matter (%)                     | 2.7        | 1.0   | 0.6  | 1.0  | –      | 0.8      | 0.8  | 0.7  | 1.1  | 0.9    |
| Chlorophyll (μg L <sup>-1</sup> )      | 10.5       | 3.8   | 16.6 | 8.1  | –      | 4.6      | 4.3  | 4.6  | 4.7  | 4.3    |
| TOC (%)                                | 9.3        | 1.7   | 1.7  | 1.7  | –      | 2.3      | 1.7  | 1.7  | 1.7  | 1.8    |
| TN (%)                                 | 0.6        | 0.2   | 0.1  | 0.2  | –      | 0.3      | 0.2  | 0.2  | 0.2  | 0.3    |
| <b>South River</b>                     |            |       |      |      |        |          |      |      |      |        |
| Water temperature (°C)                 |            |       |      |      |        |          |      |      |      |        |
| September 2010                         | 22.8       | 23.2  | 25.2 | 25.0 | –      | 22.9     | 23.6 | 24.6 | 24.6 | 24.0   |
| May/June 2011                          | 21.2       | 22.0  | 23.5 | 23.1 | –      | 21.3     | 21.5 | 22.7 | 22.8 | 21.2   |
| Salinity                               |            |       |      |      |        |          |      |      |      |        |
| September 2010                         | 12.6       | 12.5  | 12.4 | 12.4 | –      | 12.6     | 12.6 | 12.4 | 12.4 | 12.6   |
| May/June 2011                          | 2.9        | 2.9   | 3.0  | 3.0  | –      | 2.9      | 2.9  | 3.0  | 3.0  | 2.9    |
| Turbidity (NTU)                        |            |       |      |      |        |          |      |      |      |        |
| September 2010                         | 6.7        | 14.6  | 13.1 | 15.4 | –      | 8.8      | 14.0 | 8.8  | 8.3  | 12.6   |
| May/June 2011                          | 12.1       | 25.2  | 15.9 | 13.5 | –      | 12.3     | 20.2 | 12.5 | 13.9 | 12.5   |
| Dissolved oxygen (mg L <sup>-1</sup> ) |            |       |      |      |        |          |      |      |      |        |
| September 2010                         | 7.8        | 5.8   | 7.4  | 7.3  | –      | 7.9      | 8.0  | 6.7  | 6.8  | 8.6    |
| May/June 2011                          | 8.6        | 6.5   | 9.4  | 10.3 | –      | 8.4      | 6.8  | 9.6  | 10.3 | 8.1    |
| Sand + gravel (%)                      | 95.6       | 91.1  | 94.5 | 94.8 | –      | 89.7     | 83.7 | 94.9 | 96.1 | 91.0   |
| Silt + clay (%)                        | 4.4        | 8.9   | 5.5  | 5.2  | –      | 10.3     | 16.3 | 5.1  | 4.0  | 9.0    |
| Organic matter (%)                     | 0.6        | 0.9   | 0.5  | 0.5  | –      | 1.0      | 1.6  | 0.8  | 0.7  | 0.8    |
| Chlorophyll (μg L <sup>-1</sup> )      | 36.4       | 23.6  | 34.1 | 39.2 | –      | 37.4     | 24.9 | 27.6 | 26.6 | 41.1   |
| TOC (%)                                | 5.9        | 1.7   | 1.7  | 1.7  | –      | 1.7      | 1.8  | 1.7  | 2.0  | 1.7    |
| TN (%)                                 | 0.5        | 0.1   | 0.2  | 0.2  | –      | 0.0      | 0.0  | 0.0  | 0.0  | 0.2    |

Fig. 5). For the East River, a pattern of declining intertidal infaunal abundance and biomass occurred among shoreline habitats with *marsh, flats > sill, gap > riprap*. Intertidal infauna were absent at riprap shorelines because rock and filter cloth, by design, completely cover existing intertidal habitat. Infaunal abundance and biomass in the intertidal were lower in early summer 2011 than late summer 2010 at all shoreline habitats, but relative abundance and biomass among shorelines were similar (no significant interaction, Supplementary Table 1). Infaunal abundance and biomass were consistently greater in the subtidal than in the intertidal for both rivers (Supplementary Table 2).

Average infaunal taxonomic distinctness (diversity) varied by shoreline habitat in the intertidal of the East River with *marsh, flats > sill, gap > riprap*, but was similar in the intertidal of the South River (Supplementary Table 1, Fig. 6). Overall, average taxonomic distinctness was similar between rivers and lower in the intertidal ( $19.6 \pm 3.8$  SD) than subtidal ( $57.5 \pm 3.5$  SD) zones. Subtidal infaunal abundance, biomass, and diversity were similar among shoreline habitats for both rivers (Supplementary Table 3 and Fig. 7).

Rock habitat (marsh-sill and riprap) supported relatively high epifaunal abundance and biomass (Fig. 5). A pattern of declining epifaunal abundance and biomass occurred among shoreline habitats of both rivers with *sill, riprap > marsh > flats, gap* (Supplementary Table 1 and Fig. 5). There was no difference between years for the East River, but the South River had reduced epifaunal biomass and abundance in 2011 (Supplementary Table 1).

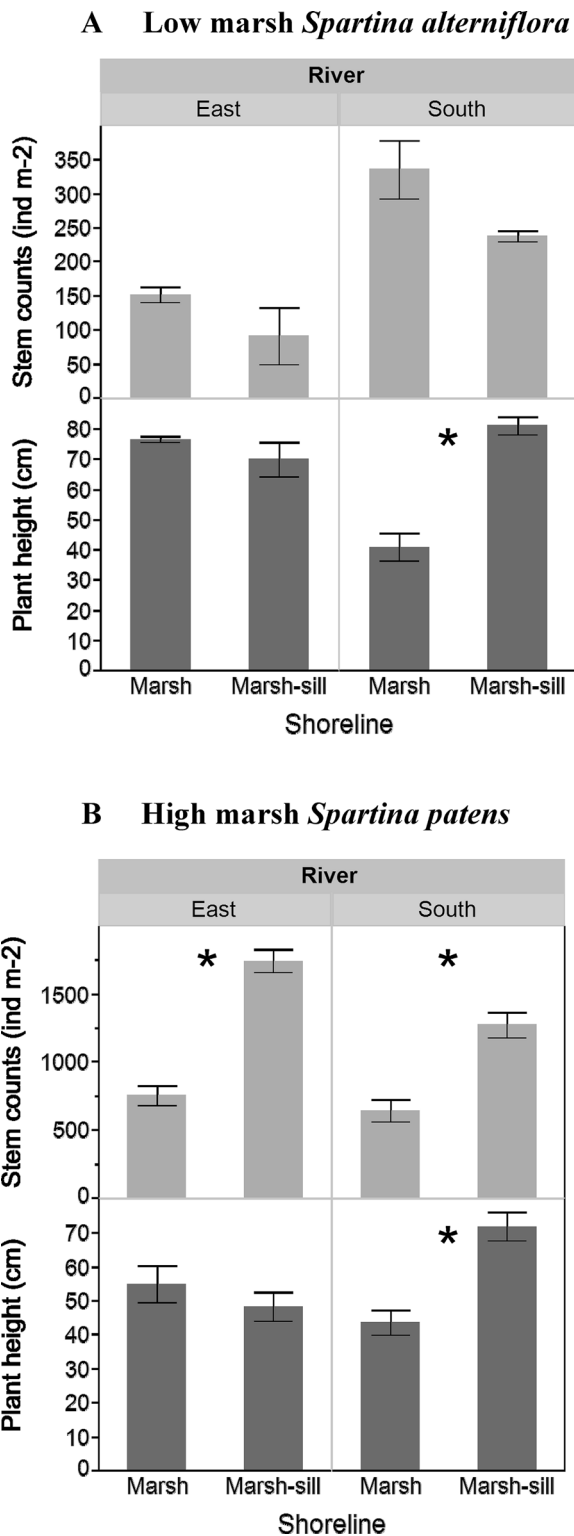
### 3.4. Infaunal community structure

Multivariate PERMANOVA tests of infaunal species abundance and biomass similarities showed differences in species composition by shoreline habitat and year in the intertidal for both rivers (Supplementary Table 4). In the subtidal, species abundance and biomass differed among shoreline habitats on the East River but not the South River, and both rivers had species differences between years. Differences between years were because of lower species abundance and biomass in early summer 2011 than in late summer 2010. Natural marsh community composition (species abundance and biomass) differed from marsh-sill habitats in the intertidal of both rivers. In the subtidal, species composition patterns varied between rivers, with species differences between marsh-sill and natural marshes in the East River, but not the South River (Supplementary Table 4).

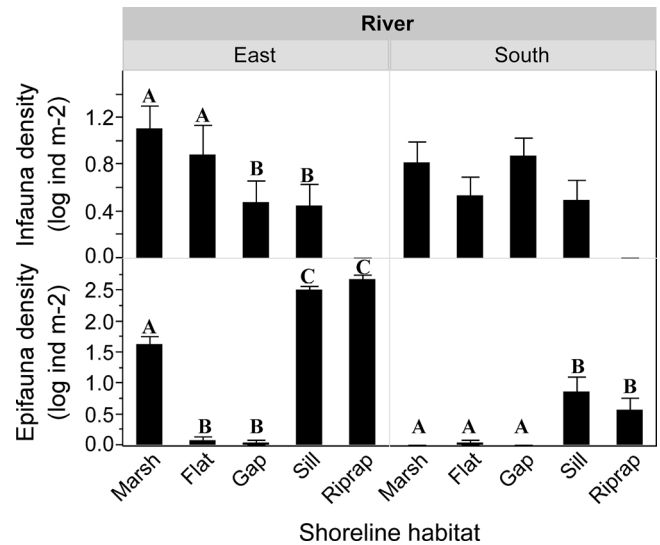
### 3.5. Feeding structure among habitats

For both rivers, natural wetlands (marsh and flats) supported a greater biomass of deposit feeders than marsh-sill habitats in the intertidal zone (Supplementary Table 1 and Fig. 8). In the intertidal, the greatest total biomass of infaunal suspension/interface-feeders occurred in natural wetlands in the East River while shoreline habitats of the South River had similar biomass. Similar infaunal feeding guild structure was present in the subtidal of all shoreline habitats

including the subtidal offshore of the marsh-sill (Supplementary Table 4). Epifaunal suspension-feeder total biomass was greatest at sill and riprap sites followed by natural marsh for both rivers (Supplementary Table 1). All subtidal habitats (marsh, flat, riprap),

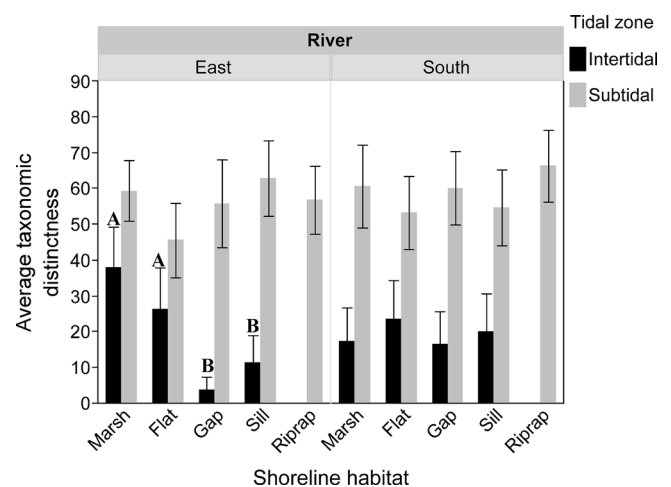


**Fig. 4.** Marsh plant characteristics. In sill marshes, low marsh plant (*Spartina alterniflora*) (A) density ( $\pm 1$  SE) was similar and height was similar or greater than natural marsh. In sill marshes, high marsh plant (*S. patens*) (B) density was greater and plant height was similar or greater than natural marshes. Asterisks indicate significant difference ( $P < 0.05$ ).



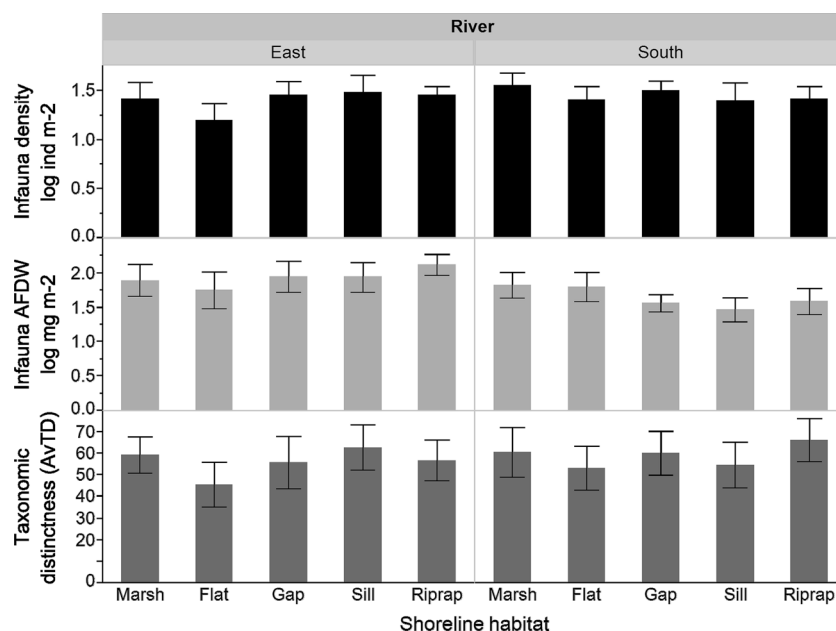
**Fig. 5.** Intertidal infaunal and epifaunal abundance. Infaunal total abundance ( $\pm 1$  SE) and biomass (not shown) were reduced at marsh-sill habitats (sill, gap) (upper panel) compared to natural wetlands (marsh, flat) in the East River but similar in the South River. Infauna were absent at riprap sites because the rock by design covers the intertidal. For each subestuary,  $n = 60$  infaunal samples (6 replicates in 2010 and 9 replicates in 2011 for each of the 4 habitats (marsh, flats, gap, and sill)). Epifaunal abundance ( $\pm 1$  SE) and biomass (not shown) were highest at sites with hard structure (sill, riprap) in both rivers (lower panel). For each subestuary,  $n = 75$  epifaunal samples (6 replicates in 2010 and 9 replicates in 2011 for each of the 5 habitats (marsh, flats, gap, sill, riprap)). Different letters indicate statistical differences ( $P < 0.05$ ).

which represent habitat for potential conversion to intertidal marsh-sill habitat, had greater biomass of suspension/interface and deposit feeders than the intertidal marsh-sill habitats (Supplementary Table 5 and Fig. 8). Supplementary Table 6 lists the species at each site and the feeding role species were assigned.



**Fig. 6.** Infaunal diversity. Infaunal diversity (mean taxonomic distinctness  $\pm 1$  SE) was reduced at intertidal marsh-sill habitats (sill, gap) compared to natural wetlands (marsh, flat) in the East River but similar in the South River. Subtidal diversity was similar among all shoreline habitats. Diversity in the intertidal was consistently lower than the subtidal for all habitats. For each subestuary,  $n = 60$  intertidal samples and  $n = 75$  subtidal samples (6 replicates in 2010 and 9 replicates in 2011 for each of the 5 habitats (marsh, flats, gap, sill, riprap)). No intertidal samples of infauna were collected for riprap which by design covers the intertidal.





**Fig. 7.** Subtidal infaunal abundance, biomass, and diversity. Total abundance, biomass (AFDW), and diversity ( $\pm 1$  SE) of infauna in the subtidal were similar among shoreline habitats in both rivers. For each subestuary,  $n = 75$  samples (6 replicates in 2010 and 9 replicates in 2011 for each of the 5 habitats (marsh, flats, gap, sill, riprap)).

## 4. Discussion

### 4.1. Ecological attributes of natural shorelines and marsh-sills

Established trajectories of function in created marshes suggest that primary producers should be functionally equivalent to natural marshes within 5 years (Craft et al., 1999); results from this study were similar suggesting that the trajectories developed for extensive created marshes may be applicable to the created fringe marshes behind the marsh sill. Marsh plant stem height, and to a lesser extent stem density, can be used as a surrogate of aboveground biomass and *Spartina* production with stem height (and production) increasing with the age of a constructed marsh (Craft et al., 2003). In this study, marsh plant communities (stem height and density) in marsh-sills 2 and 8 years old were similar to or greater than natural fringing marshes. Stem height in our fringe marshes may be reaching equivalence sooner than is occurring in extensive created *Spartina* marshes where one study reports < 10 years for stem height to become equivalent (Craft et al., 2003). *S. alterniflora* stem densities (which can affect sediment trapping capacity; Gleason et al., 1979) were similar in natural (132–464 stems m<sup>-2</sup>) and marsh-sills (180–368 stems m<sup>-2</sup>) and within previously reported ranges of stem density for natural and created marshes in the region. On average, natural fringe marshes have *S. alterniflora* densities of 143 stems m<sup>-2</sup> in Virginia [M. Mitchell, unpublished data]. Overall, stem density in created fringe marshes (e.g., in Virginia, 100–620 stems m<sup>-2</sup>, Hardaway et al., 1984; in North Carolina, 164 stems m<sup>-2</sup>, Currin et al., 2008) are similar to those reported in more extensive created marshes (e.g., in Virginia, 290 stems m<sup>-2</sup>, Havens et al., 1995; in South Carolina, 199–257 stems m<sup>-2</sup>, LaSalle et al., 1991).

Other attributes of wetland structure, such as benthic infauna, develop more slowly than the plant community. Known factors influencing benthic organisms in Chesapeake Bay are TOC and TN, sediment composition, and salinity (Boesch, 1977; Snelgrove and Butman, 1994). Sediment organic matter can be a significant source of recycled nutrients for water column productivity during

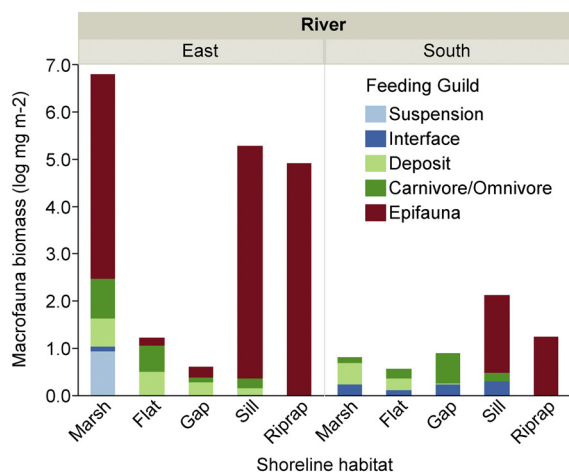
decomposition and is a source of food and energy. Created marshes may require in excess of 5–10 years to attain comparable biogeochemical processes such as organic matter and nutrient accumulation as natural wetlands (Craft et al., 2003). Constructed salt marshes <20–25 years old may have lower epifaunal and infaunal densities and fewer subsurface deposit-feeders than natural marshes, possibly due to low soil organic matter content which may limit infauna colonization in recently constructed marshes (Moy and Levin, 1991; Sacco et al., 1994; Levin et al., 1996; Scatolini and Zedler, 1996). The percentage of organic carbon in the sediments of our natural marshes (6 and 9%) fell within the range of most salt marshes along the Atlantic and Gulf Coasts (6–15%) (Craft et al., 1999), while our marsh-sills, which had coarse intertidal sediments as a result of the sand fill used during construction, had relatively low TOC (2%) and TN and a differing infaunal community structure than natural marshes. This suggests that the marsh-sills had not attained equivalent ecosystem functioning as natural marshes. However, marsh-sill sediments had total organic carbon to nitrogen ratios <20, which indicates that microbial needs are satisfied and sufficient N is available for plant uptake (Tisdale et al., 1985). Additional sampling of older marsh-sills (>20 years old) and tracking of marsh-sill functioning over time will help determine if and when functional equivalence to natural wetland can be attained by these systems.

### 4.2. Ecological tradeoffs of converting existing intertidal habitat to marsh-sill

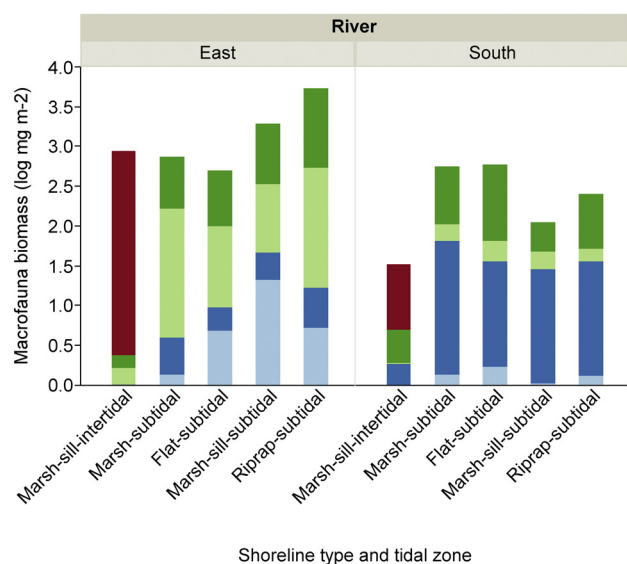
Marsh-sill intertidal macrobenthic assemblages were comprised of a combination of taxa observed in association with the unvegetated tidal flats and riprap revetment. Riprap shorelines excluded infauna but provided substrate for epifaunal colonization. Rock sills supported similar densities of epifauna as riprap revetments and, because the sill is placed slightly offshore, the intertidal is preserved allowing recruitment of infauna.

Variation in epifaunal and infaunal communities between marsh-sills and natural wetlands at our study sites suggests that

### A Intertidal feeding guild composition



### B Feeding guild composition of marsh-sill intertidal versus subtidal habitats



**Fig. 8.** Biomass of macrobenthos by feeding guilds among shorelines. Comparative biomass (AFDW) of macrobenthic feeding guilds between converted marsh-sill intertidal habitat and habitats that are commonly converted to marsh-sills: (A) natural and hardened intertidal habitats (marsh, flat, riprap), and (B) shallow subtidal habitats. Macrobenthic assemblages in marsh-sill intertidal included a combination of taxa observed in association with intertidal flats (infauna) and on riprap (epifauna). Epifaunal species were all suspension-feeders. Marsh-sill intertidal habitats have comparatively less deposit-feeders than natural marsh intertidal and shallow subtidal bottoms.

an ecological tradeoff may be occurring with marsh-sill placement. Water filtration capabilities along marsh-sills may be comparable or enhanced relative to flats (which are frequently the habitats converted to marsh-sills) due to the (a) similar biomass of suspension-feeding infauna, and (b) introduction of native filter-feeding epifauna (e.g., oysters, mussels, barnacles). While numerous factors influence the rate of seston removal by suspension feeders including water flow, depth, seasonality, and phytoplankton biomass and size (Newell and Langdon, 1996; Fulford et al., 2007), in general, increased biomass of suspension-feeders leads to increased filtration and may reduce eutrophication and pollution, and improve water clarity (Officer et al., 1982; Cohen et al., 1984; Wilkinson et al., 1996; Neubauer, 2000; Newell et al., 2005). However, the relatively low biomass of deposit-feeding

infauna observed along marsh-sills, suggests possible reductions in sediment-mixing (bioturbation) with undetermined consequences on nutrient-cycling and oxygenation. Deposit-feeders influence sediment impacting, oxygenation, and structure (Rhoads and Young, 1970; Whitlatch, 1980; Grant et al., 1982; Diaz and Schaffner, 1990), detrital and fecal matter recycling (Snelgrove, 1998), and carbon, nitrogen, phosphorus and contaminant cycling (Diaz and Schaffner, 1990). Benthic macrofauna are also a source of food for many organisms and have been estimated to directly support approximately 50% of the fish production in the Chesapeake Bay (Baird and Ulanowicz, 1989) and a fisheries yield of 27,500 Mt of carbon (Diaz and Schaffner, 1990). Further, infauna were reported to be the most significant prey group for four dominant epibenthic predators in a Chesapeake Bay subestuary (Hines et al., 1990).

The regional ecological consequences of converting existing soft-bottom intertidal habitat to artificial rocky shore are currently unknown. Some studies have documented the colonization of epifauna on hard shoreline stabilization structures (e.g., riprap, offshore breakwaters) and noted differences from natural shoreline assemblages (Chapman, 2003), but few have focused on low, free standing, stone sills (essentially reduced versions of offshore breakwaters placed closer to shore) that comprise marsh-sills (Wong et al., 2011). Moschella et al. (2005) suggested that in Europe low-crested coastal defense structures (offshore breakwaters) were poor substitutes for natural rocky shores, with less diverse and abundant epifauna on artificial structures, possibly due to less structural complexity and higher disturbance than natural shorelines. In Chesapeake Bay and along much of the Mid-Atlantic and Southeast coast, soft-bottom habitat dominates and natural rocky shorelines are rare; therefore, introduction of artificial rocky shorelines may enhance recruitment of species that are normally limited by the availability of suitable substrate including native and non-native species. In our study, only native species colonized structures, however, others have noted that artificial structure may serve as an entry point for invasive or non-local fauna (Davis et al., 2002). Artificial structures can also provide a means for dispersal of native and non-native species, potentially acting as stepping stones within a matrix of unsuitable habitat and facilitating the expansion of opportunistic and invasive species (Airolidi and Bulleri, 2011; Mineur et al., 2012). Climate change may increase the occurrence, establishment, and rate of dispersal of non-natives on artificial structures as species ranges shift in conjunction with warming waters.

Offshore macrobenthic infaunal community structure was similar among all habitats in each subestuary (although taxa varied), suggesting that any adverse effect of marsh-sill construction on infauna appears to be localized to the project footprint or over a short time scale (<2 years) prior to recovery. Recovery rates of benthos to anthropogenic disturbance (e.g., dredging, sediment disposal) can depend on the type of sediment, system size, the composition of nearby communities, the amount of sediment removed, and salinity (Newell et al., 1998). In estuarine systems, recovery rates have been reported to be within 2 years (Newell et al., 1998; Bolam and Rees, 2003; Schaffner, 2010); therefore, it is possible that the benthos were temporarily disturbed during living shoreline implementation and recovered prior to our sampling. For example, the opportunistic phoronid worm have been reported as recovering quickly and in greater abundance following the disturbance of a major tropical storm while other infauna species remained absent or in reduced numbers (Boesch et al., 1976). The presence of relatively high numbers of phoronids offshore of the East River marsh-sill site may be indicative of recovery and irruption of an opportunistic species following shoreline disturbance.

#### 4.3. Ecological tradeoffs of converting existing subtidal habitat to marsh-sill

Although this study was limited in scope, results indicate potential marsh-sill impacts that are worthy of consideration by the management community. Replacing shallow subtidal with marsh-sill habitat may significantly reduce infaunal abundance, biomass, and diversity as well as change the community structure at the location of sill construction. There is a strong spatial gradient moving from subtidal to intertidal environments for macrobenthic communities primarily due to varying exposure and inundation during tidal cycles (Kneib, 1984; Ysebaert et al., 2003). In general as the gradient moves toward greater environmental harshness (subtidal–intertidal), macrobenthic diversity declines (Kneib, 1984; Bazairi et al., 2003). We observed higher species abundance, biomass, and diversity in the subtidal than intertidal for all shorelines. In addition to differences in exposure during a tidal cycle, sediment characteristics may be influencing the distribution of benthic infauna. In a reversal from natural marsh patterns, marsh-sills had more coarse-grain sediment in the intertidal (11.5% gravel) than the subtidal (4.0% gravel) because of the practice of filling with coarse-grain sediment during project construction (aimed at limiting resuspension and loss of material). The increased grain-size at marsh-sills may be limiting the abundance of deposit-feeders which attain high densities in soft mud (Rhoads and Young, 1970); correspondingly, there was a relatively high abundance of deposit-feeders at our natural marsh sites which have more fine sediment. The conversion of shallow subtidal to vegetated intertidal-sill resulted in a comparative loss of infauna, while the addition of rock structure facilitated recruitment of filter feeding epifauna which may have offset some of the loss of infaunal filtration capacity. However species recruitment to sills will be highly variable and regulated by many factors such as species distribution, life history, distance from source populations, relative dominance of pioneer species, estuarine circulation patterns, physical conditions, and wave energy (for example, Sutherland and Karlson, 1977; Bushek, 1988; Barnes et al., 2010). In this study, the South River marsh-sill was located at the head of a tidal creek in a low-energy setting and was colonized only by *Balanus* species and extensive macroalgae. The East River marsh-sill was in a moderate-energy setting and near a source oyster and mussel population that facilitated recruitment by those species. While some filtration capacity was regained through newly recruited biota, the loss of sediment mixing services ascribed to infaunal deposit-feeders was not replaced.

Though any placement of a marsh-sill channelward will effectively reduce the overall area of shallow subtidal at a given site, in areas with extensive shallow waters the effect on the macrobenthic community should be minimal. In this study, shallow subtidal habitat adjacent to the marsh-sills supported similar benthic communities as shallows adjacent to other shorelines possibly due to the maintenance of sufficient offshore shallows (slopes were comparable between marsh-sills and natural marshes). In areas where shallow subtidal habitat is significantly limited, any potential adverse effect on the ecosystem may be magnified (e.g., higher proportional loss of refuge and feeding habitat for juvenile fish and invertebrates could reduce recruitment in the greater estuary). Minimizing the footprint of sill structures in these areas is recommended to mitigate potential effects on infauna. In general, a more balanced approach to marsh-sill design, which considers both the intertidal and subtidal habitat components, would be beneficial. The design of marsh-sills should consider subtidal elements and maintain a suitably shallow slope for establishment of the infaunal community that is a characteristic of shallow subtidal habitats.

#### 4.4. Marsh-sills as coastal habitat restoration

A defining element for a hybrid stabilization management option is that it not only provides erosion control, but it also provides or enhances habitat and water quality ecosystem services. Therefore, successful marsh-sill projects should meet stated ecological restoration goals, as well as their shoreline protection goal. In instances where shoreline protection is of highest import to the detriment of ecological objectives, the shoreline management approach should not be defined as a 'living shoreline'. Inaccurate categorization of these management practices, which may still be less adverse to the ecosystem than traditional armoring, diminishes the validity of restoration success characterizations of 'living shoreline' projects. Clearly stated shoreline management practice goals are necessary to select accurate measures of success to monitor prior to and after project completion. To identify structural and functional equivalence of marsh-sill restoration projects, one can in part apply performance criteria from created wetlands, such as plant growth, sediment organic carbon, organic matter, and nitrogen, and secondary productivity (i.e., macrobenthos, predators). However, additional performance metrics are needed to evaluate marsh-sill as these hybrid designs often introduce novel structural habitat (rock) to a soft-bottom system. Epifaunal community structure may be a particularly suitable measure as it is easily and inexpensively obtained. Use of multiple performance criteria in concert will create a more complete picture of shoreline functioning, and long-term monitoring will demonstrate whether marsh-sills do follow created marsh trajectories toward ecosystem equivalence.

In general, there was a demonstrative benefit in constructing a marsh-sill instead of the use of riprap revetment in terms of the presence of intertidal infauna with a diversity of ecological roles. Our data suggest that a marsh-sill may be viewed as providing a net positive ecological benefit when (i) the only alternative is traditional hardening (bulkhead, riprap), (ii) the sill is likely to be colonized by filter-feeding epifauna due to placement within the estuary, and/or (iii) the sill footprint can be minimized and shallow subtidal habitat maintained. Alternatively, a marsh-sill should be viewed more negatively in situations where the sill unnecessarily or extensively replaces existing habitat. The use of living shorelines (e.g., marsh-sill, created marsh with fiber coir log, grading and vegetative stabilization) as a form of shoreline protection has increased over the past two decades with approximately 10–14% of requested tidal shoreline permits in Virginia being some form of living shoreline in 2009–2011 (CCRM, 2012). Recent legislation in several states that promotes the use of these techniques along with public acceptance through demonstrations of project efficacy will likely further increase the presence of living shorelines, including marsh-sills, on our shorelines. To ensure the effectiveness of living shorelines and maximize potential ecological benefits, proper siting and design guidance to identify the most appropriate shoreline protection approaches for specific physical settings (e.g., shoreline orientation and morphology, fetch, shallow-water footprint, position in estuary) should be improved and made available to local decision-makers.

Because human interventions (e.g., shoreline modification) are not necessarily distributed randomly in nature but are instead implemented based on a specific criteria (e.g., exposure), there are limitations to the causal inferences that can be drawn from this study (Benedetti-Cecchi and Osio, 2007). Our sampling design allowed for a relative comparison of representative sites within the subestuary (Manly, 2009) and inferences should be limited to site-specific comparisons. However, it is also important to note that for many biological responses (e.g., epifaunal biomass and species composition, infaunal structure, *S. alterniflora* stem density) the

effect of habitat was consistent between subestuaries and relative habitat comparisons within subestuaries were consistent between years sampled. This suggests there is stronger evidence that an impact is occurring from marsh-sill placement (i.e., the effect is real and not from a confounding variable).

When creating marshes that require structural support, there should be a careful balance of minimizing the loss of existing habitats while encouraging the use of suitable structural habitat for epifaunal recruitment (e.g., oysters). Ranking of created marsh with structures in relation to natural habitats can be difficult because of potential detrimental effects of artificial structure placement. A comparison of secondary production in marsh-sill habitat, salt marsh, seagrass, tidal flats, and oyster reefs showed that marsh-sill productivity was high because of epifaunal assemblages, but the adverse effects of structures on macrobenthos in adjacent habitats may diminish these reported benefits (Wong et al., 2011). Further confounding issues are the numerous site-dependent factors that will affect the recruitment and establishment of epifauna. These factors should be considered during project planning to manage expectations of shoreline function. For example, oysters may not recruit to a given area due to unsuitable salinity or flow regime; therefore, oysters cannot always be expected to be present on a marsh-sill. However, other epifauna species may provide not only water filtration services, but also support marsh growth, and may even be incorporated into natural shoreline protection approaches (e.g., mussels integrated into created marshes supported by fiber coir logs). The continued exploration of 'living shoreline' designs that incorporate a variety of biological components will allow for a robust array of habitat restoration alternatives that may more closely reflect natural conditions and prevent erosion.

## Acknowledgments

Many thanks to K. Angstadt, D. Stanhope, P. Mason, C. Hershner, S. Killeen, K. O'Brien, M. Johnston, T. Russell, R. Chambers for their help in the field. Special thanks to J. Davis and W. Priest for assistance with site selection and to property owners Mr. Hult and Ms. Teague for access to sites and informative conversations. This research was funded by Chesapeake Bay Trust, NOAA, and Maryland Department of Environment. This paper is Contribution No. 3318 of the Virginia Institute of Marine Science, The College of William & Mary.

## Appendix A. Supplementary data

Supplementary material related to this article can be found, in the online version, at <http://dx.doi.org/10.1016/j.ecoleng.2013.10.011>.

## References

- Airolidi, L., Bulleri, F., 2011. Anthropogenic disturbance can determine the magnitude of opportunistic species responses on marine urban infrastructures. *PLoS ONE* 6, 1–9.
- Anderson, M.J., Gorley, R.N., Clarke, K.R., 2008. PERMANOVA for PRIMER: Guide to Software and Statistical Methods. PRIMER-E Ltd., Plymouth, United Kingdom, pp. 214.
- Baird, D., Ulanowicz, R.E., 1989. The seasonal dynamics of the Chesapeake Bay ecosystem. *Ecol. Monogr.* 59, 329–364.
- Barnes, B.B., Luckenbach, M.W., Kingsley-Smith, P.R., 2010. Oyster reef community interactions: the effect of resident fauna on oyster (*Crassostrea* spp.) larval recruitment. *J. Exp. Mar. Biol. Ecol.* 391, 169–177.
- Bazairi, H., Bayed, A., Glémarec, M., Hily, C., 2003. Spatial organisation of macrobenthic communities in response to environmental factors in a coastal lagoon of the NW African coast (Merja Zerga Morocco). *Oceanol. Acta* 26, 457–471.
- Benedetti-Cecchi, L., Osio, G.C., 2007. Replication and mitigation of effects of confounding variables in environmental impact assessment: effect of marinas on rocky-shore assemblages. *Mar. Ecol. Prog. Ser.* 334, 21–35.
- Berlin, A., 2008. Foraging Values of *Mulinia lateralis* and *Ischadium recurvum*: Energetics Effects of Surf Scoters Wintering in the Chesapeake Bay. University of Maryland, College Park, pp. 137 (PhD Dissertation).
- Berman, M.R., Berquist, H., Dewing, S., Glover, J., Hershner, C.H., Rudnick, T., Schatt, D.E., Skunda, K., 2000. Mathews County Shoreline Situation Report, Special Report in Applied Marine Science and Ocean Engineering No. 364. Virginia Institute of Marine Science, Gloucester Point, VA, pp. 93.
- Berman, M.R., Berquist, H., Killeen, S., Nunez, K., Rudnick, T., Schatt, D.E., Weiss, D., Reay, K., 2006. Anne Arundel County, Maryland – Shoreline Situation Report. Virginia Institute of Marine Science, Gloucester Point, VA, pp. 30.
- Bilkovic, D.M., Hershner, C., Rudnick, T., Nunez, K., Schatt, D., Killeen, S., Berman, M., 2009. Vulnerability of Shallow Tidal Water Habitats in Virginia to Climate Change. Final Report to NOAA Chesapeake Bay Office, Virginia Institute of Marine Science, Center for Coastal Resources Management, Gloucester Point, VA [http://ccrm.vims.edu/research/climate\\_change/COASTALHABITATS.FinalReport.pdf](http://ccrm.vims.edu/research/climate_change/COASTALHABITATS.FinalReport.pdf)
- Bilkovic, D.M., Roggero, M., 2008. Effects of coastal development on nearshore estuarine nekton communities. *Mar. Ecol. Prog. Ser.* 358, 27–39.
- Bilkovic, D.M., Roggero, M., Hershner, C.H., Havens, K.H., 2006. Influence of land use on macrobenthic communities in nearshore estuarine habitats. *Estuaries Coasts* 29, 1185–1195.
- Boesch, D.F., 1977. A new look at the zonation of benthos along the estuarine gradient. In: Coull, B.C. (Ed.), *Ecology of Marine Benthos*. University of South Carolina Press, Columbia, pp. 245–266.
- Boesch, D., Diaz, R., Virnstein, R., 1976. Effects of tropical storm Agnes on soft-bottom macrobenthic communities of the James and York estuaries and the lower Chesapeake Bay. *Chesapeake Sci.* 17, 246–259.
- Bolam, S.G., Rees, H.L., 2003. Minimizing impacts of maintenance dredged material disposal in the coastal environment: a habitat approach. *Environ. Manage.* 32, 171–188.
- Bozek, C.M., Burdick, D.M., 2005. Impacts of seawalls on saltmarsh plant communities in the Great Bay Estuary, New Hampshire, USA. *Wetlands Ecol. Manage.* 13, 553–568.
- Burke, D.G., Koch, E.W., Stevenson, J.C., 2005. Assessment of Hybrid Type Shore Erosion Control Projects in Maryland's Chesapeake Bay – Phases I & II. Final Report for Chesapeake Bay Trust, Annapolis, MD., pp. 112.
- Bushek, D., 1988. Settlement as a major determinant of intertidal oyster and barnacle distributions along a horizontal gradient. *J. Exp. Mar. Biol. Ecol.* 122, 1–18.
- Center for Coastal Resources Management (CCRM), 2010. Study of Tidal Shoreline Management in Virginia: Recommendations for Living Shorelines and Tidal Resources Sustainability. Report to the Governor and Virginia General Assembly in Response to Senate Joint Resolution No. 35, Gloucester Point, VA., pp. 30.
- Center for Coastal Resources Management (CCRM), 2012. VIMS Shoreline Permit Application Records, 1970–2011. Virginia Institute of Marine Science, Gloucester Point, VA <http://ccrm.vims.edu/perms/newpermits.html>
- Chambers, R., Meyerson, L.A., Saltonstall, K., 1999. Expansion of *Phragmites australis* into tidal wetlands of North America. *Aquat. Bot.* 64, 261–273.
- Chapman, M.G., 2003. Paucity of mobile species on constructed seawalls: effects of urbanization on biodiversity. *Mar. Ecol. Prog. Ser.* 264, 21–29.
- Christiansen, T., Wiberg, P.L., Milligan, T.G., 2000. Flow and sediment transport on a tidal salt marsh surface. *Estuar. Coast. Shelf Sci.* 50, 315–331.
- Clarke, K., Warwick, R., 1999. The taxonomic distinctness measure of biodiversity: weighting of step lengths between hierarchical levels. *Mar. Ecol. Prog. Ser.* 184, 21–29.
- Cohen, R.R.H., Dresler, P.V., Phillips, E.J.P., Cory, R.L., 1984. The effect of the Asiatic clam, *Corbicula fluminea*, on phytoplankton of the Potomac River, Maryland. *Limnol. Oceanogr.* 29, 170–180.
- Craft, C., Reader, J., Sacco, J.N., Broome, S.W., 1999. Twenty-five years of ecosystem development of constructed *Spartina alterniflora* (Loisel) marshes. *Ecol. Appl.* 9, 1405–1419.
- Craft, C., Megonigal, P., Broome, S.W., Stevenson, J., Freese, R., Cornell, J., Sacco, J., 2003. The pace of ecosystem development of constructed *Spartina alterniflora* marshes. *Ecol. Appl.* 13, 1417–1432.
- Curran, C.A., Delano, P.C., Valdes-Weaver, L.M., 2008. Utilization of a citizen monitoring protocol to assess the structure and function of natural and stabilized fringing salt marshes in North Carolina. *Wetlands Ecol. Manage.* 16, 97–118.
- Curran, C.A., Chappell, W.S., Deaton, A., 2010. Developing alternative shoreline armoring strategies: the living shoreline approach in North Carolina. In: Shipman, H., Dethier, M.N., Gelfenbaum, G., Fresh, K.L., Dinicola, R.S. (Eds.), *Puget Sound Shorelines and the Impacts of Armoring—Proceedings of a State of the Science Workshop*, May 2009: US Geological Survey Scientific Investigations Report 2010–5254. Reston, VA, pp. 91–102.
- Davis, J.L.D., Levin, L.A., Walther, S.M., 2002. Artificial armored shorelines: sites for open-coast species in a southern California bay. *Mar. Biol.* 140, 1249–1262.
- Davis, J.L.D., Takacs, R.L., Schnabel, R., 2006. Evaluating ecological impacts of living shorelines and shoreline habitat elements: an example from the upper western Chesapeake Bay. In: Erdle, S.Y., Davis, J.L.D., Sellner, K.G. (Eds.), *Management, Policy, Science, and Engineering of Nonstructural Erosion Control in the Chesapeake Bay*. CRC Publication No. 08-164, Gloucester Point, VA, pp. 55–61.
- Dauer, D.M., Lane, M.F., Llansó, R.J., 2002. Development of Diagnostic Approaches to Determine Sources of Anthropogenic Stress Affecting Benthic Community Condition in the Chesapeake Bay. Final Report to the US. Environmental Protection Agency., pp. 64.
- Diaz, R.J., Schaffner, L.C., 1990. The functional role of estuarine benthos. In: Haire, M., Krome, E.C. (Eds.), *Perspectives on the Chesapeake Bay*. Advances in Estuarine



- Sciences. United States Environmental Protection Agency, Gloucester Point, VA, pp. 25–56.
- Dugan, J.E., Airoldi, L., Chapman, M.G., Walker, S.J., Schlacher, T., 2011. Estuarine and coastal structures: environmental effects, a focus on shore and nearshore structures. In: Wolanski, E., McLusky, D. (Eds.), *Treatise on Estuarine and Coastal Science*. Elsevier Press, New York, pp. 17–41.
- Folk, R.L., 1980. *Petrology of Sedimentary Rocks*. Hemphill Pub. Co, Austin, TX, pp. 190.
- Fulford, R.S., Breitburg, D.L., Newell, R.I.E., Kemp, W.M., Luckenbach, M., 2007. Effects of oyster population restoration strategies on phytoplankton biomass in Chesapeake Bay: a flexible modeling approach. *Mar. Ecol. Prog. Ser.* 336, 43–61.
- Gleason, M.L., Elmer, D.A., Pien, N.C., Fisher, J.S., 1979. Effects of stem density upon sediment retention by salt marsh cord grass, *Spartina alterniflora* Loisel. *Estuaries* 2, 271–273.
- Grant, W.D., Boyer, L.F., Sanford, L.P., 1982. The effects of bioturbation on the initiation of motion of intertidal sands. *J. Mar. Res.* 40, 659–677.
- Hardaway, C.S., Thomas, G.R., Zacherle, A.W., Fowler, B.K., 1984. *Vegetative Erosion Control Project: Final Report*. Prepared for Virginia Division of Soil and Water. Technical Report. Virginia Institute of Marine Science, College of William and Mary, Gloucester Point, VA, pp. 275.
- Havens, K.J., Varnell, L.M., Bradshaw, J.G., 1995. An assessment of ecological conditions in a constructed tidal marsh and two natural reference tidal marshes in coastal Virginia. *Ecol. Eng.* 4, 117–141.
- Hines, A.H., Haddon, A.M., Wiechert, L.A., 1990. Guild structure and foraging impact of blue crabs and epibenthic fish in a subestuary of Chesapeake Bay. *Mar. Ecol. Prog. Ser.* 67, 105–126.
- Holland, A.F., Mountford, N.K., Mihursky, J.A., 1977. Temporal variation in upper bay mesohaline benthic communities. I. The 9-m mud habitat. *Chesapeake Sci.* 18, 370–378.
- Hughes, R.N., Seed, R., 1981. Size selection of mussels by the blue crab *Callinectes sapidus*: energy maximizer or time minimizer? *Mar. Ecol. Prog. Ser.* 6, 83–89.
- King, R.S., Deluca, W.V., Whigham, D.F., Marra, P.P., 2007. Threshold effects of coastal urbanization on *Phragmites australis* (common reed) abundance and foliar nitrogen in Chesapeake Bay. *Estuaries Coasts* 30, 469–481.
- King, R.S., Hines, A.H., Craige, F.D., Grap, S., 2005. Regional, watershed and local correlates of blue crab and bivalve abundances in subestuaries of Chesapeake Bay, USA. *J. Exp. Mar. Biol. Ecol.* 319, 101–116.
- Kneib, R.T., 1984. Patterns of invertebrate distribution and abundance in the intertidal salt marsh: causes and questions. *Estuaries* 7, 392–412.
- LaSalle, M.W., Landin, M.C., Sims, J.G., 1991. Evaluation of the flora and fauna of a *Spartina alterniflora* marsh established on dredged material in Winyah Bay, South Carolina. *Wetlands* 11, 191–208.
- Levin, L.A., Talley, D., Thayer, G., 1996. Succession of macrobenthos in a created salt marsh. *Mar. Ecol. Prog. Ser.* 141, 67–82.
- MacIntyre, H.L., Geider, R.J., Miller, D.C., 1996. Microphytobenthos: the ecological role of the “secret garden” of unvegetated, shallow-water marine habitats. I. Distribution, abundance and primary production. *Estuaries* 19, 186–201.
- Madsen, K.N., Nilsson, P., Sundbäck, K., 1993. The influence of benthic microalgae on the stability of a subtidal sediment. *J. Exp. Mar. Biol. Ecol.* 170, 159–177.
- Manly, B.F.J., 2009. *Statistics for Environmental Science and Management*, second ed. Chapman and Hall/CRC Press, New York, pp. 295.
- Mann, R., Southworth, M., Harding, J.M., Wesson, J.A., 2009. Population studies of the native eastern oyster, *Crassostrea virginica* (Gmelin, 1791) in the James River, Virginia, USA. *J. Shellfish Res.* 28, 193–220.
- McGranahan, G., Balk, D., Anderson, B., 2007. The rising tide: assessing the risks of climate change and human settlements in low elevation coastal zones. *Environ. Urban.* 19, 17–37.
- Middelburg, J.J., Barranguet, C., Boschker, H.T.S., Herman, P.M.J., Moens, T., Heip, C.H.R., 2000. The fate of intertidal microphytobenthos: an in situ  $^{13}\text{C}$  labeling study. *Limnol. Oceanogr.* 45, 1224–1234.
- Miller, D.C., Geider, R.J., MacIntyre, H.L., 1996. Microphytobenthos: the ecological role of the “secret garden” of unvegetated, shallow-water marine habitats. II. Role in sediment stability and shallow-water food webs. *Estuaries* 19, 202–212.
- Mineur, F., Cool, E.J., Minchin, D., Bohn, K., Macleod, A., Maggs, C.A., 2012. Changing coasts: marine aliens and artificial structures. *Oceanogr. Mar. Biol.: Annu. Rev.* 50, 189–234.
- Morgan, P.A., Burdick, D.M., Short, F.T., 2009. The functions and values of fringing salt marshes in northern New England, USA. *Estuaries Coasts* 32, 483–495.
- Morley, S.A., Toft, J.D., Hanson, K.M., 2012. Ecological effects of shoreline armoring on intertidal habitats of a Puget Sound urban estuary. *Estuaries Coasts* 35, 774–784.
- Morse, J.L., Megonigal, J.P., Walbridge, M.R., 2004. Sediment nutrient accumulation and nutrient availability in two tidal freshwater marshes along the Mattaponi River, Virginia, USA. *Biogeochemistry* 69, 175–206.
- Moschella, P.S., Abbiati, M., Åberg, P., Airoldi, L., Anderson, J.M., Bacchiocchi, F., Bulteri, F., Dinesen, G.E., Frost, M., Gacia, E., Granhag, L., Jonsson, P.R., Satta, M.P., Sundelöf, A., Thompson, R.C., Hawkins, S.J., 2005. Low-crested coastal defence structures as artificial habitats for marine life: using ecological criteria in design. *Coast. Eng.* 52, 1053–1071.
- Moy, L.D., Levin, L.A., 1991. Are *Spartina* marshes a replaceable resource? A functional approach to evaluation of marsh creation efforts. *Estuaries* 14, 1–16.
- National Research Council (NRC), 2007. *Mitigating Shore Erosion on Sheltered Coasts*. The National Academies Press, Washington, DC, pp. 188.
- Neubauer, M.T., 2000. *Demographics, Production, and Benthic–Pelagic Coupling by the Suspension Feeding Polychaete Chaetopterus pergamentaceus in the Lower Chesapeake Bay*. The College of William and Mary, Williamsburg, VA (PhD Dissertation).
- Neubauer, S.C., Anderson, I.C., Constantine, J.A., Kuehl, S.A., 2002. Sediment deposition and accretion in a Mid-Atlantic (U.S.A.) tidal freshwater marsh. *Estuar. Coast. Shelf Sci.* 54, 713–727.
- Newell, R.C., Seiderer, L.J., Hitchcock, D.R., 1998. The impact of dredging works in coastal waters: a review of the sensitivity to disturbance and subsequent recovery of biological resources on the sea bed. *Oceanogr. Mar. Biol.: Annu. Rev.* 36, 127–178.
- Newell, R.I.E., Langdon, C.J., 1996. Mechanisms and physiology of larval and adult feeding. In: Kennedy, V.S., Newell, R.I.E., Eble, A. (Eds.), *The eastern oyster, Crassostrea virginica*. Maryland Sea Grant Publication, College Park, MD, USA, pp. 185–230.
- Newell, R.I.E., Fisher, T.R., Holyoke, R.R., Cornwell, J.C., 2005. Influence of eastern oysters on nitrogen and phosphorus regeneration in Chesapeake Bay, USA. In: Dame, R., Olenin, S. (Eds.), *The Comparative Roles of Suspension Feeders in Ecosystems*. NATO Science Series: IV – Earth and Environmental Sciences, vol. 47. Springer, Netherlands, pp. 93–120.
- Officer, C.B., Smayda, T.J., Mann, R., 1982. Benthic filter feeding, a natural eutrophication control. *Mar. Ecol. Prog. Ser.* 9, 203–210.
- Peterson, M.S., Comyns, B.H., Hendon, J.R., Bond, P.J., Duff, G.A., 2000. Habitat use by early life-history stages of fishes and crustaceans along a changing estuarine landscape: differences between natural and altered shoreline sites. *Wetlands Ecol. Manage.* 8, 209–219.
- Peterson, M.S., Lowe, M.R., 2009. Implications of cumulative impacts to estuarine and marine habitat quality for fish and invertebrate resources. *Rev. Fish. Sci.* 17, 505–523.
- Rhoads, D.C., Young, D.K., 1970. The influence of deposit-feeding organisms on sediment stability and community trophic structure. *J. Mar. Res.* 28, 150–178.
- Ricciardi, A., Bourget, E., 1998. Weight-to-weight conversion factors for marine benthic macroinvertebrates. *Mar. Ecol. Prog. Ser.* 163, 245–251.
- Ruiz, G.M., Hines, A.H., Posey, M.H., 1993. Shallow water as a refuge habitat for fish and crustaceans in non-vegetated estuaries: an example from Chesapeake Bay. *Mar. Ecol. Prog. Ser.* 99, 1–16.
- Sacco, J.N., Seneca, E.D., Wentworth, T.R., 1994. Infaunal community development of artificially established salt marshes in North Carolina. *Estuaries* 17, 489–500.
- Scatolini, S.R., Zedler, J.B., 1996. Epibenthic invertebrates of natural and constructed marshes of San Diego Bay. *Wetlands* 16, 24–37.
- Schaffner, L.C., 2010. Patterns and rates of recovery of macrobenthic communities in a polyhaline temperate estuary following sediment disturbance: effects of disturbance severity and potential importance of non-local processes. *Estuaries Coasts* 33, 1300–1313.
- Seitz, R., Lipcius, R., Olmstead, N., 2006. Influence of shallow-water habitats and shoreline development on abundance, biomass, and diversity of benthic prey and predators in Chesapeake Bay. *Mar. Ecol. Prog. Ser.* 326, 11–27.
- Snelgrove, P., 1998. The biodiversity of macrofaunal organisms in marine sediments. *Biodivers. Conserv.* 7, 1123–1132.
- Snelgrove, P.V.R., Butman, C.A., 1994. Animal-sediment relationships revisited: cause versus effect. *Oceanogr. Mar. Biol.: Annu. Rev.* 32, 111–177.
- Sutherland, J.P., Karlson, R.H., 1977. Development and stability of the fouling community at Beaufort, North Carolina. *Ecol. Monogr.* 47, 425–446.
- Tisdale, S.L., Nelson, W.L., Beaton, J.D., 1985. *Soil Fertility and Fertilizers*. Macmillan Pub. Co., New York, pp. 754.
- Tyler, A.C., McGlathery, K.J., Anderson, I.C., 2003. Benthic algae control sediment-water column fluxes of organic and inorganic nitrogen compounds in a temperate lagoon. *Limnol. Oceanogr.* 48, 2125–2137.
- Underwood, G.J.C., Paterson, D.M., 1993. Seasonal changes in diatom biomass, sediment stability and biogenic stabilization in the Severn Estuary. *J. Mar. Biol. Assoc. U. K.* 73, 871–887.
- Whitlatch, R.B., 1980. Patterns of resource utilization and coexistence in marine intertidal deposit-feeding communities. *J. Mar. Res.* 38, 743–765.
- Wilkinson, S.B., Zheng, W., Allen, J.R., Fielding, N.J., Wanstall, V.C., Russell, G., Hawkins, S.J., 1996. Water quality improvements in Liverpool Docks: the role of filter feeders in algal and nutrient dynamics. *Mar. Ecol. Prog. Ser.* 137, 197–211.
- Wong, M., Peterson, C., Piehler, M., 2011. Evaluating estuarine habitats using secondary production as a proxy for food web support. *Mar. Ecol. Prog. Ser.* 440, 11–25.
- Ysebaert, T., Herman, P.M.J., Meire, P., Craeymeersch, J., Verbeek, H., Heip, C.H.R., 2003. Large-scale spatial patterns in estuaries: estuarine macrobenthic communities in the Schelde estuary, NW Europe. *Estuar. Coast. Shelf Sci.* 57, 335–355.