4-2012

Ecological and erosion protection functions of Chesapeake Bay living shorelines : Comprehensive Monitoring of Ecological and Erosion Protection Functions of Chesapeake Bay Living Shorelines (CMLS), Phase II

Donna Marie Bilkovic
Virginia Institute of Marine Science

Molly Mitchell
Virginia Institute of Marine Science

Recommended Citation

Bilkovic, D., & Mitchell, M. (2012) Ecological and erosion protection functions of Chesapeake Bay living shorelines : Comprehensive Monitoring of Ecological and Erosion Protection Functions of Chesapeake Bay Living Shorelines (CMLS), Phase II. Virginia Institute of Marine Science, College of William and Mary. https://doi.org/10.25773/dhe7-jb59

ECOLOGICAL AND EROSION PROTECTION FUNCTIONS OF CHESAPEAKE BAY LIVING SHORELINES

Final Report to Chesapeake Bay Trust, National Oceanic and Atmospheric Administration (NOAA) Restoration Center, and the Maryland Department of the Environment

Project: Comprehensive Monitoring of Ecological and Erosion Protection Functions of Chesapeake Bay Living Shorelines (CMLS), Phase II, Grant #10670

April 2012

Donna Marie Bilkovic
Molly Mitchell

Virginia Institute of Marine Science, College of William & Mary,
PO Box 1346, Gloucester Point, VA, 23062

BACKGROUND

Armoring shorelines to prevent erosion, improve access, and accommodate individual landscaping interests can result in fragmentation or loss of habitats, reduction in capacity to moderate pollutant loads delivered to coastal waters, reduction in nekton and macrobenthic integrity (Bilkovic et al. 2005, King et al. 2005, Seitz et al. 2006, Bilkovic et al. 2006, Bilkovic & Roggero 2008), increases in invasive species, such as *Phragmites australis* (Chambers et al. 1999, King et al. 2007), and disturbance of sediment budgets sustaining adjacent properties. As an alternative to traditional armoring of shorelines, shoreline protection techniques incorporating natural elements from the system are increasingly promoted as not only less harmful to the system, but also beneficial due to their ability to provide or enhance coastal ecosystem services. Yet, 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).

Living Shorelines defined

Leading to confusion, several terms have been used synonymously to represent shoreline stabilization techniques that strive to preserve or restore the natural character of the shoreline and intertidal zone. Terms include *bioengineered*, *soft*, *green*, *natural*, *non-structural* or *alternative shoreline stabilization,* as well as *living shorelines*. In addition, stabilization techniques that are labeled with these terms often differ dramatically in their approaches and potential ecosystem function. To adequately define the expected ecosystem services from these approaches, types of shoreline stabilization have to be carefully parsed out and generalizations eliminated.

For the purposes of this research, an unambiguous definition of a natural shoreline stabilization approach was extracted from existing uses to reduce confusion and the inclusion of inappropriate stabilization strategies. Natural approaches to shoreline stabilization (termed ‘living shorelines’ from this point forward) have been defined in several ways, but are typically comprised of a few common elements. Living shorelines techniques

1) use natural habitat elements (e.g. vegetation) to protect shorelines from erosion
2) do not include structures that sever natural processes and connections between riparian, intertidal and aquatic areas, such as tidal exchange, sediment movement, plant community transitions and groundwater flow
3) provide habitat and water quality ecosystem services

In sum, living shorelines are shoreline management approaches that use natural elements, such as vegetation, to protect shorelines from erosion, provide or enhance habitat and water quality ecosystem services, and preserve the natural processes and connections between riparian, intertidal and subaqueous areas.

Not all living shorelines are created alike

There are two primary types of living shoreline used in the Chesapeake Bay that fulfill the stated definition, 1) non-structural (e.g. vegetation, fill, and coir logs) and 2) hybrid (rock structure used to support vegetation growth) (**Fig. 1**). Hybrid techniques incorporate non-structural approaches for erosion control in combination with more traditional approaches; however, these are placed in a manner that do not sever the physical connection to the riparian, intertidal and subaqueous areas to qualify as living shoreline practices. In general, non-structural approaches are considered more likely to succeed in low wave energy environs, while hybrid techniques are typically applied in areas of medium to high wave energy.

Figure 1. Non-structural living shoreline marsh planting (left) and hybrid living shoreline with planted marsh and rock sill (right).

To evaluate the success of a restoration project, well-designed and cost-effective monitoring plans are required to document the relative change in ecosystem services that occur as a result of the restoration activities. Effective monitoring approaches clearly describe expected benefits from a restoration activity and develop performance measures to assess success. Monitoring data can also provide information to improve future restoration activities and designs. *Living shoreline* habitat restoration activities are typically designed to control erosion, while simultaneously enhancing estuarine habitats. Expected outcomes are shoreline protection, estuarine habitat creation in the intertidal, beach and subaqueous zones, and enhanced habitat services for fauna and flora communities. However, uncertainty remains in regards to the effectiveness of living shorelines at meeting expected ecological or engineering goals. This is in part due to the lack of empirical information about the trade-offs involved in habitat conversion (i.e. loss of subtidal habitat), and is particularly true for hybrid living shoreline projects in higher energy systems that include rock structure, such as marsh-sills (low "free standing" stone structures placed near the marsh shoreline) (**Fig. 2**).

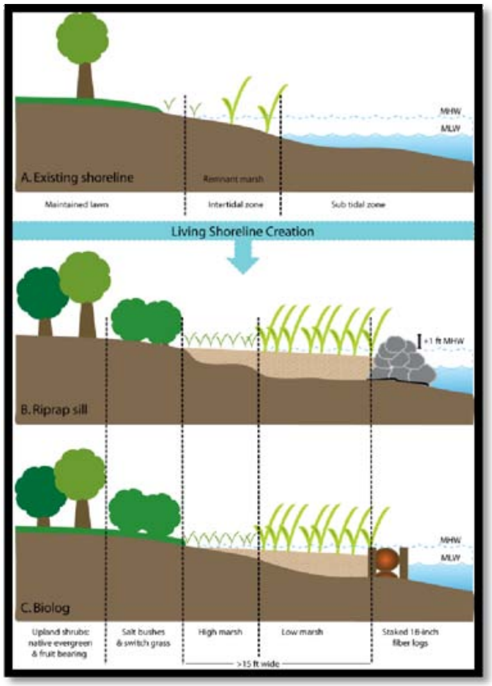


Figure 2. Depiction of a typical living shoreline treatments built channelward with conversion of existing unvegetated wetland (flats) and subaqueous (subtidal) lands to sand fill and planted marsh. These designs result in a wider intertidal area and a change in elevation, sediment type and plant usage but better maintain the upland-water connection compared to riprap revetments and bulkheads.

To begin to address the uncertainty, a comprehensive monitoring protocol for living shorelines was developed and implemented from 2010 – 2012 with funding from Chesapeake Bay Trust, NOAA Habitat Conservation and Maryland Department of the Environment to examine key coastal management questions. The study was structured to empirically evaluate habitat conversion trade-offs of living shoreline placement as well as their effectiveness as erosion protection.

STUDY QUESTIONS

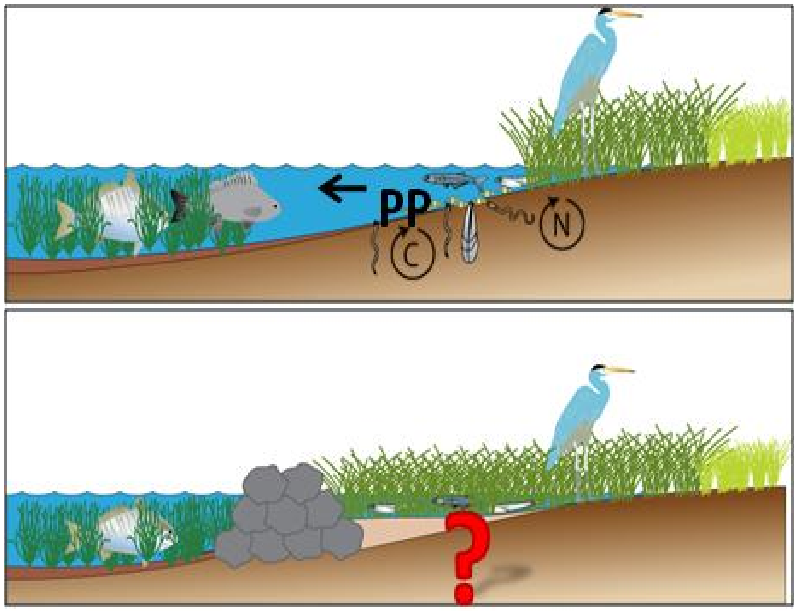
1. Do marsh-sill shorelines provide similar ecosystem services as natural shorelines?

2. What are the ecological tradeoffs of converting existing intertidal habitat to hard structure (sill, riprap)?

3. What are the ecological tradeoffs of converting existing subtidal habitat to vegetated marsh-sill habitat?

4. Are macrobenthic communities in the shallow subtidal habitats offshore of marsh-sills similar to those offshore of natural shorelines?

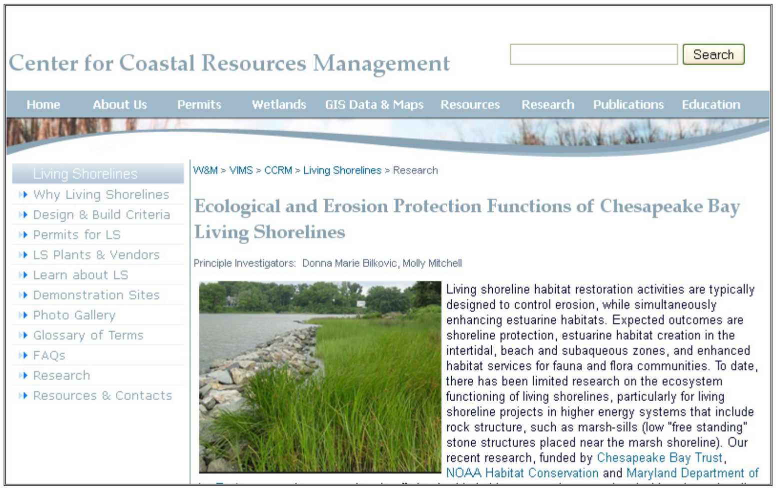
5. Do marsh-sills provide comparable erosion protection to natural and/or riprap revetment shorelines?



Results from the first sampling phase (Bilkovic & Mitchell 2011, Ecological and Erosion Protection Functions of Chesapeake Bay Living Shorelines, Phase I) suggest that living shorelines provided shoreline stabilization, and may be following established created wetland trajectories (i.e. equivalence after 1-5 yrs for primary producers & 5-25 yrs for benthic infauna particularly subsurface deposit feeders) (e.g., Craft et al. 2003). Marsh plant communities were comparable to natural marshes in terms of density and plant height, which is representative of aboveground biomass. Following major storms Hurricane Irene and Tropical Storm Lee, elevation surveys of the marsh-sill living shorelines suggest that the shorelines were protected and the sills appear to be "hot spots" for the collection of sediment, and capable of retaining the sediment during storm events.

Other attributes of wetland structure, such as benthic infauna, develop more slowly than the plant community. Constructed salt marshes less than 20-25 years may have lower epifauna and infauna densities and fewer subsurface deposit feeders than in natural marshes, possibly due to low soil organic matter content which may limit infauna colonization in recently constructed marshes. The age of the living shoreline should be considered during evaluation of ecosystem functioning. The surveyed living shorelines in this study were between 2 and 8 years of age and did not yet support equivalent infauna as natural marshes.

The placement of living shorelines involves the conversion of existing unvegetated intertidal and subtidal bottoms to a vegetated intertidal and/or rock sill. These existing shallow habitats support highly productive benthic microalgal communities that contribute significantly to primary production in estuaries, support higher tropic levels and maintain sediment stability. The unvegetated intertidal and shallow subtidal also provide refuge and feeding habitat for juvenile fish and invertebrates. A website with study results from Phase I of the project is located at: http://ccrm.vims.edu/livingshorelines/eco-erosion-function/index.html



Ecological and Erosion Protection Functions of Chesapeake Bay Living Shorelines

Principle Investigators: Donna Marie Bilkovic, Molly Mitchell

Living shoreline habitat restoration activities are typically designed to control erosion, while simultaneously enhancing estuarine habitats. Expected outcomes are shoreline protection, estuarine habitat creation in the intertidal, beach and subaqueous zones, and enhanced habitat services for fauna and flora communities. To date, there has been limited research on the ecosystem functioning of living shorelines, particularly for living shoreline projects in higher energy systems that include rock structure, such as marsh-sills (low "free standing" stone structures placed near the marsh shoreline). Our recent research, funded by Chesapeake Bay Trust, NOAA Habitat Conservation and Maryland Department of

The purpose of the second phase of the study was to continue and expand the comprehensive monitoring program that was initiated in July 2010 and funded through the Chesapeake Bay Trust. The rationale for expansion of monitoring components was derived from observations during sampling events in 2010 and the need for temporal community comparisons due to potential variation in recruitment. Both the abundance and distribution of benthic macrofauna is influenced by seasonal cycles; therefore, the Phase II sampling was conducted much earlier in the summer. Seasonal sampling furthered our knowledge of the varying patterns of recruitment of certain major taxa and variability in primary production; helping to determine the spatial and temporal scales at which monitoring is required to effectively determine living shoreline ecological functioning.

In addition, research suggests that properly placed wetlands (i.e. living shorelines) are a cost-effective method of reducing anthropogenic non-point source pollution (e.g. Tobias et al. 2001; Deegan et al. 2007). Sediment nutrients were assessed during the second phase of the study to begin a comparative evaluation of the nitrogen and phosphorus removal potential of living shorelines. Comparative data will strengthen our understanding of ecosystem services provided by these restoration activities and may provide further justification for the expansion of their application across the Bay watershed. This is particularly timely as a Chesapeake Bay watershed-wide TMDL is currently being established with specific guidelines for each watershed state and district for implementation and accountability for nutrient and sediment reductions.

Macrobenthos

Benthic macrofauna are important components of estuarine and coastal ecosystems, because they are critical links between primary producers, organic matter sources (e.g. phytoplankton, benthic algae, detritus) and fish & crustaceans. They make ideal indicators of habitat quality in that they respond quickly to impairments, are mostly sedentary thus reflect local conditions, and provide many ecosystem services to maintain good water and sediment quality.

- Infauna are animals that live in the substrate of a body of water. They include *polychaetes, oligochaetes, bivalves, and crustaceans*.
- Epifauna are animals living on or just above the substrate. They may be firmly attached (sessile), relatively sedentary, or highly motile. Common Chesapeake Bay examples include *oysters, mussels, barnacles, snails, sponges & sea squirts.*

Benthic macrofauna have been linked with a variety of ecosystem services, relating to their feeding strategies, habitat alterations and production.

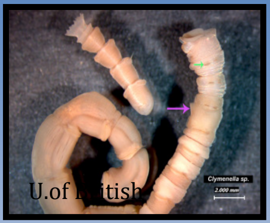
- Suspension feeders (primarily bivalves & annelids) filter suspended material and pollutants from the water column, reducing eutrophication, improving water clarity and shuttling organic matter from a pelagic to benthic food web (e.g., Cohen et al. 1984, Newell 1988, Neubauer 2000).
- Deposit feeders and tube builders (primarily annelids & crustaceans) bioturbate the sediment which may increase sediment oxygenation, impact sediment stability, and change sediment structure (e.g., Rhoads & Young 1970, Whitlatch 1980, Grant et al. 1982, Diaz & Schaffner 1990). They can also affect carbon and nitrogen cycling pathways by recycling detrital and fecal matter back into the food chain (Snelgrove 1998).
- Macrobenthos are a source of food for many organisms (including a direct link to human consumption for some species). They have been estimated to directly support approximately 50% of the fish production in the Chesapeake Bay (Baird & Ulanowicz 1989) and a fisheries yield of 27,500 metric tons of carbon (Diaz & Schaffner 1990).

Macrobenthos: Ecosystem Service Providers

<u>Deep Deposit feeders</u>

- Ingest sediment & digest associated bacteria, microalgae & organic matter
- Bioturbate sediment – increase oxygenation & nutrient cycling

Infauna



Clymenella torquata
Bamboo worm

Marenzelleria viridis
Red-gilled mud worm

<u>Suspension/filter feeders</u>

- Feed on algae & detrital particles suspended in the water
- Filter water, improve clarity

Infauna

Tagelus plebeius
Stout razor clam

Epifauna

Crassostrea virginica
Eastern oyster

Geukensia demissa
Atlantic ribbed mussel

METHODS

Site Selection & Survey Design

We conducted a paired-site comparison of *marsh-sill living shoreline* versus *natural and hardened shoreline* types (natural marsh, unvegetated flats, and riprap revetment) with the same three locations in Maryland and Virginia used for the Phase I study (**Fig.3**). Sites were selected with several criteria including comparable salinity and energy regimes, available adjacent comparative habitats, sediment characteristics, age of project, and accessibility. For each marsh-sill location, adjacent habitats of natural marsh, unvegetated flats and riprap revetment were selected for comparative surveys that also met certain criteria including: minimum length of shoreline (≥ 30 m contiguous shoreline condition), and similar sediment type, salinity and energy regime, and depth profiles to marsh-sills (**Table 1**). Sampling occurred in September 2010 (Phase I) and May–June 2011 (Phase II) to capture varying patterns of recruitment for benthic taxa.

Two locations surveyed have marsh-sills of similar designs (East and South rivers) but varying tidal gap sizes between sills. The Severn River living shoreline differed as it was a created marsh with a submerged continuous subtidal sill and a biolog. Because the sill is subtidal, the biolog is the dominant intertidal feature and this site will be referred to as the "Biolog" site in the results.


East River –Marsh–sill small tidal gaps


South River –Marsh–sill large tidal gaps


Severn River – Created Marsh with Biolog

At each shoreline site, we applied similar protocols followed during late summer 2010 surveys, and augmented monitoring components with additional sediment nutrient and Chlorophyll *a* analyses. Nine randomly selected transects were surveyed for each habitat type (i.e., sill, gap, biolog, intertidal flat, riprap, marsh). Transects followed perpendicular to the shore from intertidal to subtidal (~1–2 m depth) zones (For an example, see **Fig. 4**). Living shoreline (marsh-sill and biolog) locations had a total of 18 transects to ensure that both sill and gap habitats were assessed adequately (i.e., 9 transects per habitat type), all other shoreline types were comprised of contiguous habitat and thus had 9 transects. On site, transects were flagged based on GPS coordinates along the shore and a previously assigned random direction was followed for each transect from the intertidal to the subtidal zone. At each sample site, ecological attributes were measured in intertidal and subtidal habitats to evaluate ecosystem service provision by living shorelines (**Table 2**).

Table 1. Shoreline site characteristics.

	East Living Shl	East Marsh	East Flat	East Riprap	South Living Shl	South Marsh	South Flat	South Riprap	Severn Living Shl	Severn Marsh	Severn Flat	Severn Riprap
Site Location	East River, VA	East River, VA	East River, VA	East River, VA	Almhouse Ck, South River, MD	Glebe Bay, South River, MD	Glebe Bay, South River, MD	Almhouse Ck, South River, MD	College Ck, Severn River, MD	Weems Ck, Severn River, MD	College Ck, Severn River, MD	Weems Ck, Severn River, MD
Site Length (m)	256	73	61	91	244	61	30	70	207	37	30	73
Riparian Land Use	Residential	Residential; Lawn; Trees	Residential; Lawn; Trees	Residential; Lawn	Residential; Lawn; Road	Forested	Residential	Residential	Riparian buffer planted; Lawn; College	Residential	Residential; Forested along shoreline	Residential; Lawn; woody vegetation
Wave energy	Moderate	Moderate	Moderate	Moderate	Low	Moderate	Low	Moderate	Low	Low	Low	Low
Widest fetch (NM)	1-5	1-5	1.2	1-5	<1	1.2	<1	1.3	<1	<1	<1	<1
Orientation	SW	SW	W	SW	NW	NW	NW	SE	NW	W	E	W-NW
Avg Slope %	1.3	2.0	2.7	6.9	5.6	3.4	1.9	9.9	15.5	11.5	6.8	20.7
Structure Length (m)	256	-	-	91	122	-	-	70	270	-	-	73
Build Date	2003-04	-	-	-	2008	-	-	~2009	2006	-	-	-
Structure description	Gapped sill with 3 sills & 3 small 8' gaps	-	-	Continuous riprap revetment	Gapped sill with 5 sills & equidistant sill/gap pattern	-	-	Continuous riprap revetment	Biolog with fully submerged continuous low sill ~3-4' offshore	-	-	Continuous riprap revetment
Marsh Length (m)	256	73	-	-	122	61	-	-	207	37	-	-
Ave low marsh width (m)	3.8	3.2	-	-	6.1	1.3	-	-	5.3	1.8	-	-
Ave high marsh width (m)	6.7	18.3	-	-	4.3	13.8	-	-	8.5	13.3	-	-

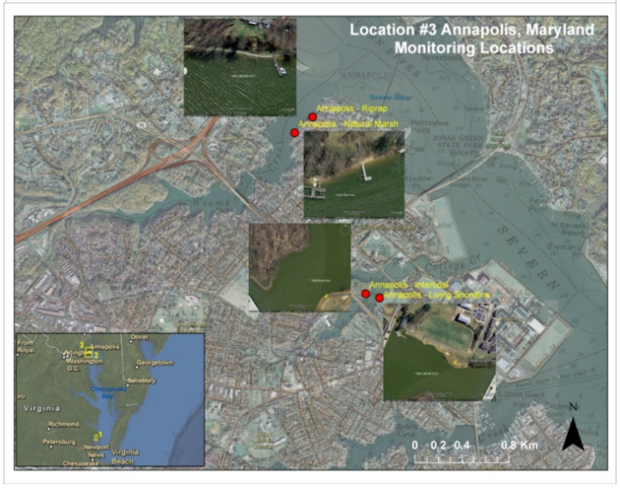


Figure 3.
Location #1: East River, Mobjack Bay in Mathews, Virginia. The marsh-sill was built in 2003-04.

Location #2: South River, Almshouse Creek in London Towne, Maryland. The marsh-sill was built in 2008.

Location #3: Severn River, College Creek, Annapolis, Maryland. The marsh and biolog/submerged sill were constructed in 2005.

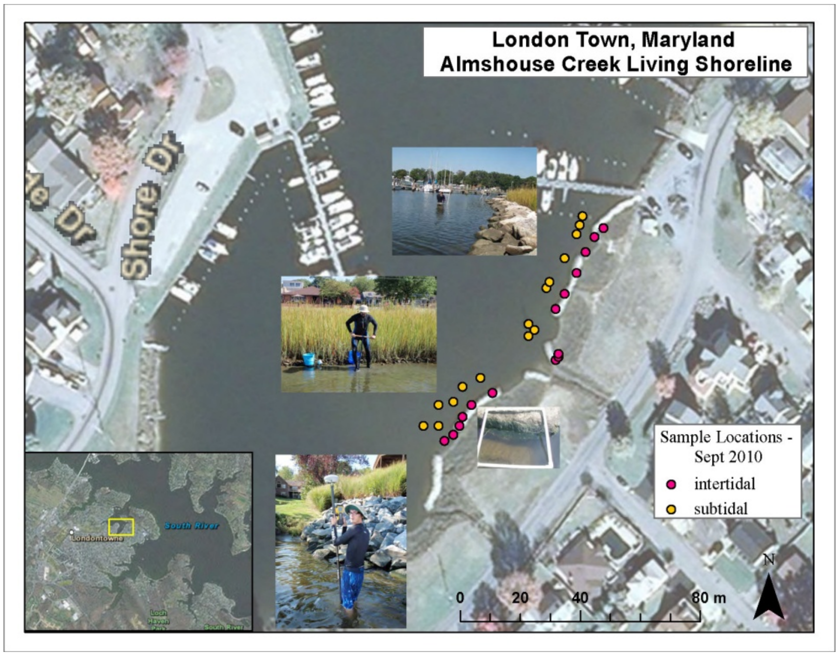


Figure 4. Example transect layout from 2010 study (15 total transects: 6 along sills, 6 within gaps and 3 near the central outfall – in 2011 6 additional sill and gap transects were added) within intertidal and subtidal zones of the South River marsh-sill shoreline. Along each transect (shore – subtidal), macrobenthos, water quality, vegetation and sediment were sampled.

Site Characteristics

We evaluated physical site characteristics onsite and remotely: slope, relative wave energy, fetch, orientation, structure length and riparian land use. We measured the slope, the distance from the shoreline to water depths of at least 1m MLW, at 3-6 transects per site, as well as with elevation data obtained during shoreline profiles at 6 sites (**see Shoreline Survey below for details**). Fetch, shoreline orientation and structure length were determined in GIS, and riparian land use and wave energy assessed onsite. In low and high marsh zones in 2011**,** we measured marsh vegetation stem count, species composition, and plant height of the 3 tallest stems within 0.25 m^2 quadrats placed randomly at 3-6 transects per site. We measured average marsh width (the distance from shoreline that water travels) at 3-6 transects per site and with aerial photography for larger marsh extents.

Physicochemical Measures

Concurrently with macrofaunal sampling, we measured physical variables including water temperature, salinity, dissolved oxygen, turbidity, water depth, and sediment grain size and organic matter that may influence benthic faunal distribution and abundance. During each sampling event, we used a hand-held YSI sonde to record dissolved oxygen, salinity, conductivity, pH, turbidity, temperature and chlorophyll *a*. Additional water samples were taken for lab verification of chlorophyll *a* readings. We collected sediment cores (15 cm depth) near macrobenthos sample locations at 3-6 cross-shore transects per site and determined grain-size, sediment nutrients and organic matter 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).

Macrobenthos Survey – Infauna and Epifauna

At each shoreline type and tidal zone, we took deep core samples (30 cm depth, 10-cm diameter) to capture shallow and deep-dwelling infaunal benthos and sediment was sieved on a 0.5-mm mesh. A modification to the benthic infaunal sampling was implemented because similar faunal communities were observed in shallow (15 cm height) and deep (30 cm height) cores in 2010 samples. In 2011, we used only the larger deep cores which allowed us to increase the number of replicates (we added 3-6 transects per site) while reducing the total number of cores taken because more area can be surveyed with the larger core. All samples were sorted and macrobenthos were identified down to the lowest practical taxonomic unit (generally species). Specimens were then dried to a constant weight (typically for 48 h) at 60°C and ashed at 550°C for 4 h to obtain ash weight. Bivalves were ashed separately from other infauna. The bivalves were shucked prior to ashing to remove additional weight of the periostracum. When there were too few of a taxon in a sample to determine AFDM, length-weight (L-W) regressions (bivalves) or mean individual weight values (annelids, arthropods) from previous studies were utilized to estimate biomass. Total abundance (number of $\text{individuals}\cdot m^{-2}$) and biomass ($mg\cdot m^{-2}$) for each site were estimated. We estimated the diversity with a taxonomic distinctness metric which has numerous advantages over traditional diversity measures such as species richness, including

- Describes phylogenetic diversity & is more closely linked to functional diversity
- Robust to variation in sampling effort and number of species
- Responsive to environmental degradation whilst being relatively insensitive to major habitat differences

- It can utilize only simple species lists (Presence/Absence data) (Clarke and Warwick 1999).

We sampled epifauna within the intertidal zone of each site, concurrently with infauna sampling. For each transect, we counted the number of each epifaunal species present within a 0.25-m^2 quadrat. Epifauna biomass was estimated with published L-W relationships. For oysters, we applied an average shell length (80.5mm; representing the average size of age 2 oysters) to an oyster biomass equation from Mann et al. (2009). For mussels, an average size of 8 cm (*Geukensia demissa* ribbed mussels) and 36 mm (*Ischadium recurvum* hooked mussels) were applied to L-W regressions (Berlin 2008, Hughes & Seed 1981). For barnacles the mean AFDM for two size classes (small and medium-large) were determined in the lab and used to estimate biomass.

In total, samples collected from all 12 shorelines included

- 243 infauna cores
- 135 epifaunal counts
- 81 sediment cores

Comparative statistical analyses

The paired site design utilized was essentially a paired control-treatment with spatial (shoreline type and watershed) components that was analyzed for differences to address posed ecological questions:

- Q1: To evaluate the ecological equivalence among tested shoreline types, *intertidal* and *subtidal* faunal community metrics (e.g. abundance, biomass, diversity) and vegetation measures were independently compared.
- Q2: To evaluate the benthic production trade-off from conversion of *intertidal* to hard structure (i.e. sill, riprap), epifaunal and infaunal community metrics were compared.
- Q3: To evaluate functional changes in converted subtidal (subaqueous) bottom, *subtidal* faunal community metrics associated with natural wetlands or riprap were examined in relation to *intertidal* (converted) marsh-sill habitats.
- Q4: To evaluate the influence of shoreline type (structure or natural) on offshore *subtidal* fauna, *subtidal* faunal community metrics were compared between structured (sill or riprap) and non-structural shorelines.

Comparative analyses was completed with generalized linear models fit to a negative binomial log-link distribution to accommodate count data which can be highly non-normal and overdispersed and sequential sidak adjustment for multiple comparisons was applied. Main model factors were River (South, Severn, East), Shoreline type (marsh, flat, sill, gap, biolog, riprap) and Year (2010, 2011).

Shoreline Surveys

- Q5: To evaluate whether marsh-sills provide comparable erosion protection to natural and/or riprap revetment shorelines we conducted high resolution shoreline profiles before and after major storm events and evaluated shoreline change.

We used an integrated GPS surveying system with application in coastal zone environments to conduct shore and nearshore surveys. To set site control and acquire shore data, we used a

Trimble R8 GNSS Model 2 Real Time Kinematic (RTK) GPS System. The RTK uses Global Positioning technology to quickly establish vertical position in the National Spatial Reference Frame with approximately +/- 2mm of accuracy. The high speed microprocessor in the Trimble R8 GNSS receiver enables precise position estimation, even in challenging environments as is often the case in the coastal zone. In addition, we used a Trimble 5600 Robotic Total Station to acquire nearshore data. Surveys encompassed several elements including structure dimensions, shoreline position and profiles from landward of the shoreline/structure to below MLW (~ -2 feet MLW contour).

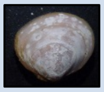
During September 2010, baseline surveys were completed at two monitoring locations, 1 in Maryland (South River, London Towne) and 1 in Virginia (East River, Mathews). At each monitoring location, the shoreline types: 1) living shoreline, 2) riprap revetment and 3) natural marsh were profiled. Horizontal and vertical controls were established by obtaining coordinates through a long static observation on each site (~ 4 hours). Surveys were repeated in May 2011 and Sept/Oct 2011 following major storm events (Hurricane Irene and the remnants of Tropical Storm Lee in late Aug-early Sept 2011). An additional living shoreline (marsh-sill) was surveyed in May and Oct 2011 in the South River adjacent and upriver of the surveyed riprap site. Vertical precision ranged between 5 and 13 mm and horizontal precision was 3 to 9 mm. All survey data were incorporated into GIS format for change analysis between survey events (pre and post storms).

Contour and cut-and-fill analysis

ArcGIS 9.3 was used to study changes in shoreline profiles and erosion patterns at the shorelines. Survey elevation data for each time frame were converted to point feature classes for use in ArcGIS. Digital elevation models (DEMs) were created for each location and time period with 3D Analyst to create TIN models of each set of data. The TINs were converted to DEMs (rasters) with a linear interpolation method. The Spatial Analyst extension was used to create zero elevation contour line from the DEMs. These zero contour lines were used to examine trends in the shoreline.

Volumetric change of each site and time frame was done with 3D Analysts Cut/Fill tool. The DEMs for the two time frames of interest were compared for areas where the elevation had increased or decreased. The Cut/Fill tool creates a raster image showing areas of net gain (deposition), net loss (erosion) and no change. Total volumetric change for each site was calculated from the raster attribute table with the Statistics tool to sum all the volume changes in the study area. Negative changes indicate net gain and positive changes indicate net loss. Total volumetric change was standardized to the Area (also calculated from the raster attribute table with the Statistics tool) to allow relative comparisons between sites.

Table 2. Ecosystem functions characterized during shoreline studies

Ecosystem Function	Ecosystem Service	Measurement
Sediment trapping, wave attenuation	Shoreline stabilization	Profiles – before & after major storm events 
Primary production *support of food webs*	Fisheries production	Stem counts, plant height, diversity measures 
Habitat *support of food webs*	Fisheries production	Infauna abundance, biomass & diversity 
Nutrient & Sediment filtration; Carbon cycling; Bioturbation	Water quality improvement	Epifauna & infauna abundance, biomass & diversity 
Sediment composition & organic matter *support of food webs*	Fisheries production & shoreline stabilization	Sediment cores – OM, Total N, P, OC and grain size 

RESULTS

1. Do living shorelines provide similar ecosystem services as natural shorelines?

Yes and No.

In created marshes, most ecological attributes reportedly follow a predictable trajectory towards structural/function equivalence to natural marshes. Within 5-15 years, primary producers and macrobenthic communities typically reached equivalence, while organic carbon and nitrogen accumulation may require in excess of 25 years (Craft et al. 2003). Our living shoreline sites ranged from 2 to 8 years of age, and if following created marsh trajectories may have reached equivalence for some ecological attributes and not others. It is possible that those attributes that are not equivalent may reach equivalence at a later date.

Living shorelines surveyed supported similar marsh plant communities in terms of composition, abundance and height. Sediment organic matter and total organic carbon to nitrogen ratios were not equivalent to natural wetlands.

Living shoreline intertidal habitat supported a lower abundance, biomass and diversity of infauna than natural wetlands (marsh & flats), but was an improvement from riprap structure which effectively eliminates intertidal habitat and infauna. Subtidal habitat of all shoreline types supported similar infauna abundance, biomass and diversity. The created marsh living shoreline on the Severn River displayed dramatic interannual variability with infauna communities that were similar to natural wetlands in 2010 and an absence of intertidal infauna in 2011. This reversal may have been driven from extensive creosote seepage from the old bulkhead due to coastal flooding in 2011.

Primary production

In natural and living shoreline planted marshes, the predominant species were *Spartina alterniflora* (low marsh) and *Spartina patens* (high marsh). 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).

Low marsh plant density (stem count · m^{-2}) was similar between living shoreline (biolog: 284 ± 82; Sill 204 ± 59) and natural marsh (233 ± 55) sites ($X^2 = 0.2$, $p = 0.98$). High marsh plant density was similar or higher in living shorelines (biolog 398 ± 115; sill 1395 ± 403) as compared to natural marshes (391 ± 92) ($X^2 = 10.2$, $p = 0.02$) (**Fig. 5 upper panel**). Plant height was similar in both low and high marsh between living shoreline (Low marsh: biolog 58.33, sill 61.4 cm; High marsh: biolog 83.3, sill 58.1cm) and natural marshes (Low marsh: 61.3; High marsh: 76.8 cm) (low marsh: X^2=0.7, p = 0.9; high marsh: X^2=2.9, p=0.4) (**Fig. 5 lower panel**).

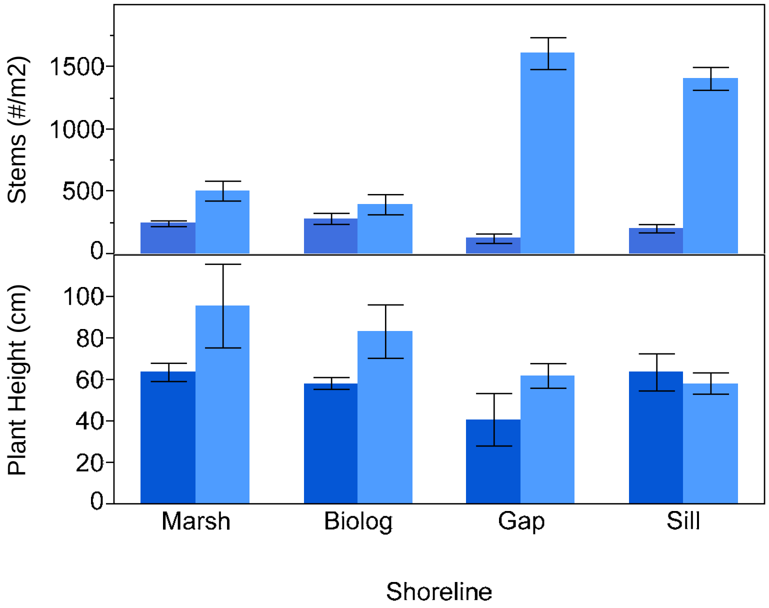


Figure 5. Living shorelines had similar or higher plant abundance in both the low and high marsh zones than natural marsh sites. Plant height was comparable between living shorelines and natural marshes in both the low and high marsh zones.

Habitat Provision

Known fundamental factors influencing benthic organisms in Chesapeake Bay are total organic carbon and total nitrogen, sediment composition (i.e., grain size) and salinity (e.g., Boesch 1977, Snelgrove & 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. Sediments at the living shorelines, which are more than 2 but less than 8 years old, do not yet reflect organic carbon content of the natural shorelines and thus may not be supporting similar habitat functions (**Fig. 6**). Total organic carbon to nitrogen ratios less than 20 indicate that microbial needs are satisfied and sufficient N is available for plant uptake (Tisdale et al. 1985) and along all shorelines surveyed this was the case. Created marshes may require in excess of 5 to10 years to attain comparable biogeochemical processes such as organic matter and nutrient accumulation as natural wetlands (Craft et al. 2003). Sediment grain-size within the intertidal varied between living shoreline sites and natural wetlands with larger grain-size at living shorelines (**Fig 7**). Physicochemical parameters dissolved oxygen, water temperature, salinity, pH, and turbidity were similar between paired living shorelines and natural wetlands (**Table 3**). Poor water quality can be characterized by chlorophyll *a* levels that are frequently above the threshold concentration (Buchanan et al. 2005; Lacouture et al. 2006). Chesapeake Bay thresholds are ≤ 20.9 ug/L for oligohaline waters, which include the South and Severn rivers, and ≤ 6.2 ug/L for mesohaline waters, which include the East River. Chlorophyll levels exceeded the threshold in South and Severn rivers at all sites, while thresholds were exceeded only for the marsh and living shoreline sites in the East River. Although, chlorophyll in the South and Severn living shoreline sites was relatively low compared to marsh and riprap sites (**Fig 8**).

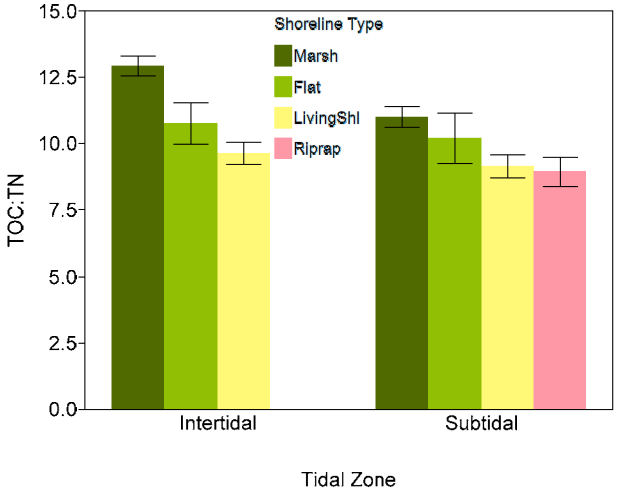


Figure 6. Total organic carbon to total nitrogen ratios along living shorelines were lower than natural wetlands. Ratios less than 20 indicate that microbial needs are satisfied and sufficient N is available for plant uptake (Tisdale et al. 1985) and along all shorelines surveyed this was the case

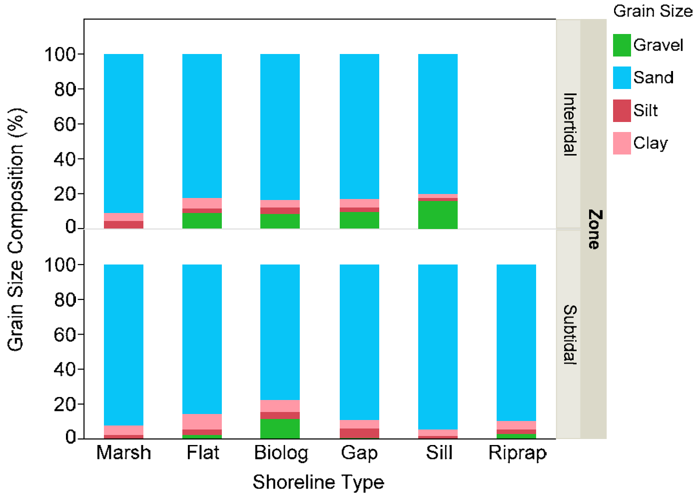


Figure 7. Living shoreline sites had a greater percentage of larger grain-size sediments (i.e., gravel) than natural marshes in the intertidal (7.8% vs. 0.09%). This is not unexpected because shoreline restoration projects involve the placement of clean large-grained sand/gravel.

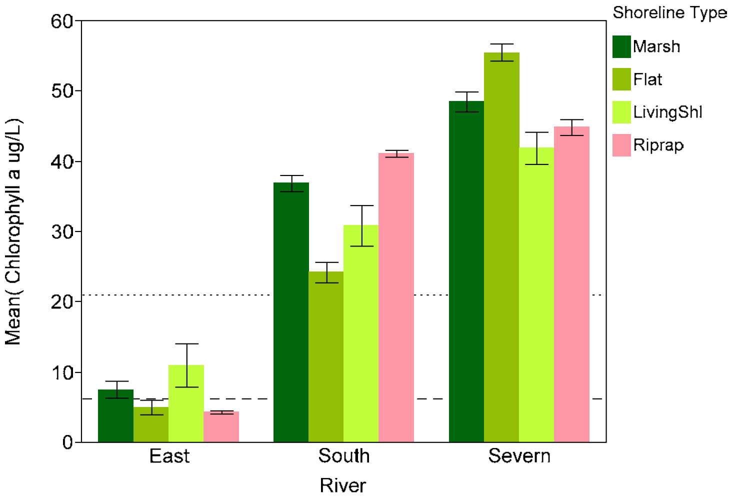


Figure 8. Poor water quality can be characterized by chlorophyll *a* levels that are frequently above the threshold concentration. Chesapeake Bay thresholds are indicated on the graphic for oligohaline waters (≤ 20.9 ug/L), which include the South and Severn rivers, and mesohaline waters (≤ 6.2 ug/L) of the East River. Chl *a* levels exceeded the threshold in South and Severn rivers at all sites, while thresholds were exceeded only for the marsh and living shoreline sites in the East River. Chl *a* in the South and Severn living shorelines was relatively low compared to marsh and riprap sites.

Table 3. Mean water quality values during sampling events in Sept 2010 & May–June 2011

River	Shoreline	Tidal zone	DO mg/L		Water temp°C		Salinity		pH		Turbidity NTU	
			2010	2011	2010	2011	2010	2011	2010	2011	2010	2011
East	LivingShl	intertidal	6.8	6.9	25.5	26.1	22.0	12.9	8.0	7.9	11.8	79.8
East	LivingShl	subtidal	7.4	9.2	25.6	29.4	22.3	14.3	8.1	8.4	12.4	7.6
East	Marsh	intertidal	6.5	7.2	26.2	28.3	22.1	14.7	7.9	8.0	18.9	37.1
East	Marsh	subtidal	6.4	7.2	26.0	27.3	22.0	14.6	8.0	8.1	19.2	12.9
East	Flat	intertidal	7.3	7.5	24.4	26.7	22.1	11.2	8.1	8.1	19.2	137.4
East	Flat	subtidal	7.3	7.2	24.9	28.4	22.0	14.2	8.1	8.2	13.5	11.1
East	Riprap	subtidal	7.9	9.2	27.0	28.0	22.0	14.8	8.1	8.3	24.2	8.9
Severn	LivingShl	intertidal	6.9	9.6	23.9	20.6	12.9	2.5	7.8	8.6	17.5	20.9
Severn	LivingShl	subtidal	6.7	10.4	23.6	21.1	12.9	2.5	7.8	8.7	23.4	28.7
Severn	Marsh	intertidal	10.2	11.0	25.5	20.9	12.3	2.8	8.2	8.8	11.4	10.2
Severn	Marsh	subtidal	10.3	11.1	25.4	21.0	12.4	2.8	8.2	8.8	12.0	12.9
Severn	Flat	intertidal	7.8	10.8	24.9	21.0	12.7	2.5	7.9	8.8	11.1	32.5
Severn	Flat	subtidal	7.3	10.7	24.5	21.1	13.0	2.5	7.9	8.8	37.1	15.9
Severn	Riprap	subtidal	8.1	11.0	24.5	23.3	12.4	2.8	8.0	8.8	25.2	25.2
South	LivingShl	intertidal	7.4	9.9	25.1	23.3	11.6	3.0	7.7	8.9	13.1	15.9
South	LivingShl	subtidal	6.8	9.9	24.6	22.7	12.4	3.0	7.8	9.0	8.8	12.5
South	Marsh	intertidal	7.8	8.6	22.8	21.2	12.6	2.9	8.1	8.7	6.7	12.1
South	Marsh	subtidal	7.9	8.4	22.9	21.3	12.6	2.9	8.1	8.7	8.8	12.3
South	Flat	intertidal	5.8	6.5	23.2	22.0	12.5	2.9	7.5	7.8	14.6	25.2
South	Flat	subtidal	8.0	6.8	23.6	21.5	12.6	2.9	8.0	8.1	14.0	20.2
South	Riprap	subtidal	8.6	8.1	24.0	21.2	12.6	2.9	8.1	8.7	12.6	12.5

Habitat Provision: Macrobenthos Communities

Macrobenthos abundance & biomass

Marsh-sill and riprap intertidal habitat supported a lower abundance and biomass of infauna than natural wetlands. Subtidal habitat of all shoreline types supported similar infauna abundance and biomass. The Severn River deviated from marsh-sill infauna and epifauna patterns of the South and East rivers due to the absence of an exposed rock sill.

Infauna

Infauna abundance and biomass was lower within intertidal sill habitats compared to natural wetlands (X^2 = 13.4 (abundance), 75.5 (biomass), $p<0.0001$, **Fig. 9, upper panel**). A pattern of declining intertidal infauna abundance and biomass occurred among shoreline habitats with *Marsh, Flats > Sill, Gap, Biolog > Riprap*. Infauna were absent at riprap shorelines because the rock completely covers existing intertidal habitat. Infauna abundance and biomass in the intertidal was lower in early summer 2011 than late summer 2010 at all shoreline types, but relative abundance and biomass among shorelines was similar. The Severn River living shoreline (i.e. created marsh with biolog) had the highest interannual variability with relatively high abundance in 2010 to an absence of intertidal infauna in 2011. This is possibly due to intense coastal flooding and release of high amounts of creosote from the remnant bulkhead– that occurred prior to and during sampling – reducing recruitment success.

Subtidal infauna abundance was similar among shoreline types (X^2 = 6.8, p = 0.2) and biomass was similar among shorelines with the exception of flats exhibiting higher biomass than the biolog site (X^2 = 36.7, $p<0.0001$). Infauna abundance and biomass in the subtidal were consistently higher than in the intertidal (X^2 = 252.9, 492.2, $p<0.0001$).

Epifauna

Rock habitat (marsh-sill & riprap revetment) supported relatively high epifauna abundance and biomass (**Fig. 9, bottom panel**). A pattern of declining epifauna abundance and biomass occurred among shoreline habitats with *Sill & Riprap >> Marsh > Flats, Gap, Biolog* (X^2 =267.0, 396.1, $p<0.0001$). The rivers did vary with higher average abundance and biomass of epifauna along East River marsh, sill and riprap shorelines as compared to other rivers (X^2 = 448.0, 300.8, $p<0.0001$). Predominant epifauna at the East River living shoreline and riprap were eastern oyster (*Crassostrea virginica*), hooked mussels (*Ischadium recurvum*) and barnacle species; the natural marsh was comprised of oysters and Atlantic ribbed mussels (*Geukensia demissa*). Within the South River, the only epifauna species observed were barnacles at the living shoreline and riprap sites. Epifauna within the Severn River consisted of barnacles only at the riprap site.

Variation in epifauna and infauna communities between living shorelines and natural wetlands suggest that an ecological trade-off may be occurring with marsh-sill placement. Increasing epifauna which were predominantly filter feeders may enhance water filtration on site; however, concomitant declines in infauna could indicate a decline in sediment bioturbation and associated nutrient cycling depending on the species or species groups that are being misplaced.

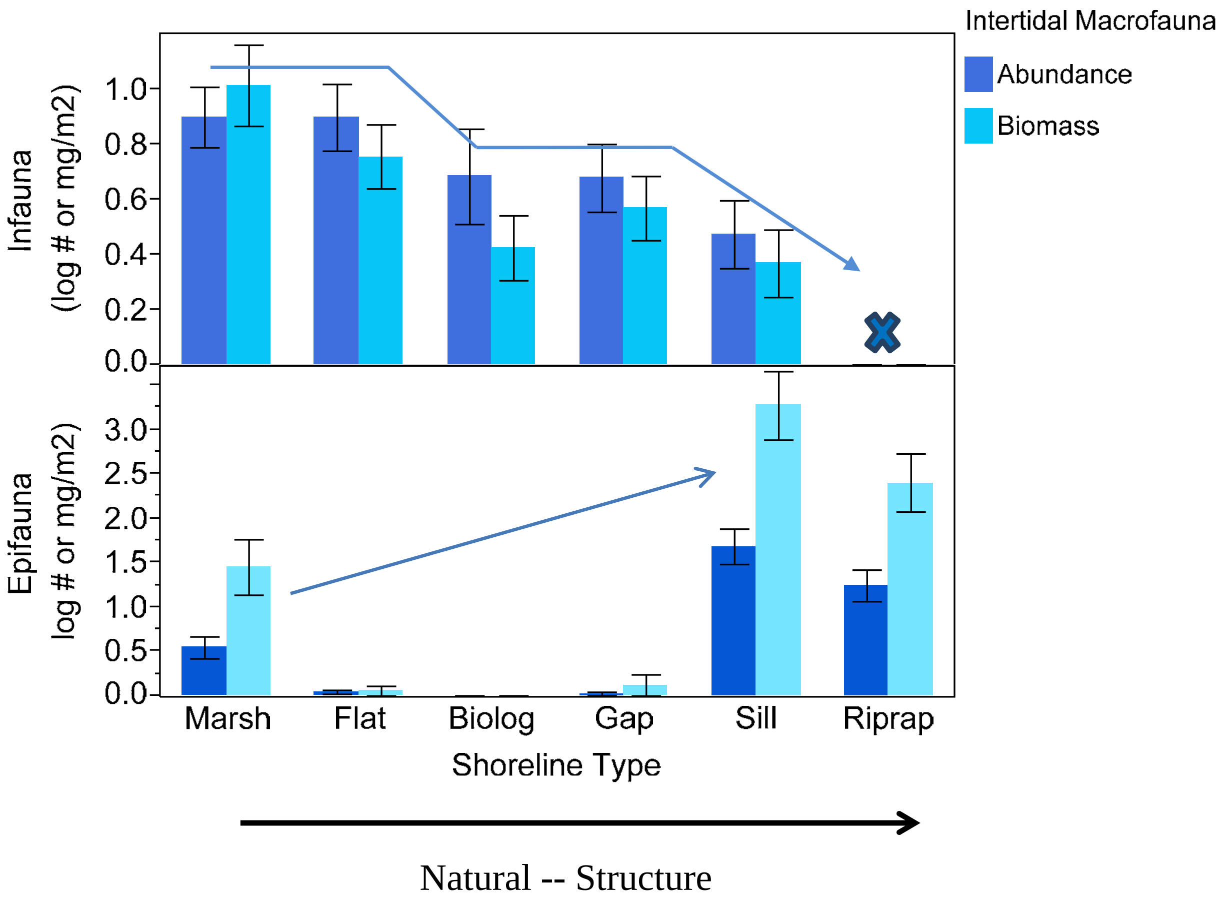


Figure 9. Infauna abundance and biomass were reduced at living shoreline habitats (sill, gap and biolog) (upper panel) compared to natural wetlands (marsh, flat). Epifauna abundance and biomass were highest at sites with hard structure (sill, riprap) (lower panel).

Macrobenthos Diversity

Marsh-sill infauna diversity was similar to natural wetlands in the intertidal and subtidal zones. Overall, average taxonomic distinctness was lower in the intertidal (17.6 ± 2.3) than subtidal (65.6 ± 2.5) zones (**Fig. 10**).

Infauna taxonomic distinctness (biodiversity) varied by shoreline type (3-factor ANOVA, F=6.9, p<0.0001) and tidal zone (F=54.0, p<0.0001), but was similar among rivers (F=2.1, p=0.1) in 2010. There was an interaction between shoreline type and tidal zone: natural wetlands (marsh, flat) exhibited similar diversity between zones while riprap and living shorelines were less diverse in intertidal than subtidal zones. In 2011, this trend changed slightly due to relatively low abundances in intertidal habitats with living shorelines supporting comparable diversity to natural marshes. Infauna taxonomic distinctness varied by shoreline type in the intertidal with *Marsh, Flats, Sill, Gap, Biolog* > *Riprap* (F=9.3, p<0.0001), but was similar in the subtidal (F=0.2, p=0.978). When combining years, diversity was similar between living shoreline habitat (biolog, gap & sill) and natural wetlands in both the intertidal and subtidal. Some caution should be applied to data interpretation due to the high observed variability of infauna in intertidal zones.

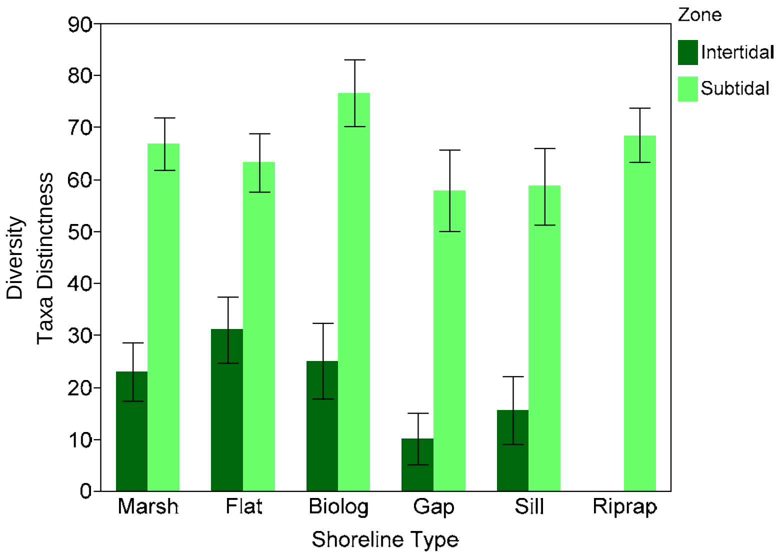


Figure 10. Diversity (mean taxonomic distinctness in 2010 and 2011) was similar between living shoreline habitat (biolog, gap & sill) and natural wetlands in both the intertidal and subtidal. Diversity in the intertidal (17.6 ± 2.3) was consistently lower than the subtidal (65.6 ± 2.5) for all habitats.

2. What are the ecological tradeoffs of converting existing intertidal habitat to hard structure (sill, riprap)?

Living shoreline (marsh-sill) macrobenthos communities were comprised of a combination of taxa observed in association with the unvegetated flats and riprap revetment (**Fig. 11**). Riprap revetment shorelines only supported epifauna and intertidal flats supported a mix of deposit-feeders, suspension-feeders and carnivore/omnivore infauna with minimal epifauna present. Natural wetlands (marsh & flats) consistently had a greater biomass of deposit feeders than living shorelines (X^2= 60.5, p < 0.0001). Suspension feeders had the greatest biomass in natural marshes, but sill and unvegetated flats were similar: *marsh >> sill, flat > gap, biolog > riprap* (X^2 = 215.5, p < 0.0001). Epifauna (filter feeders) biomass was highest at sill and riprap sites followed by natural marsh: (*sill, riprap > marsh > flat, gap, biolog*) (X^2 = 396.1, p <0.0001).

There may be comparable or enhanced water filtration capabilities in the living shorelines as flats (which are frequently the habitats converted to living shorelines) due to the *a)* comparable biomass of suspension-feeding infauna, and *b)* possible introduction of new filter-feeding epifauna (e.g., oysters, barnacles). However, the reduction of deep deposit-feeding infauna observed along marsh-sill living shorelines, suggests possible reductions in sediment-mixing (bioturbation) with undetermined consequences on nutrient cycling and oxygenation.

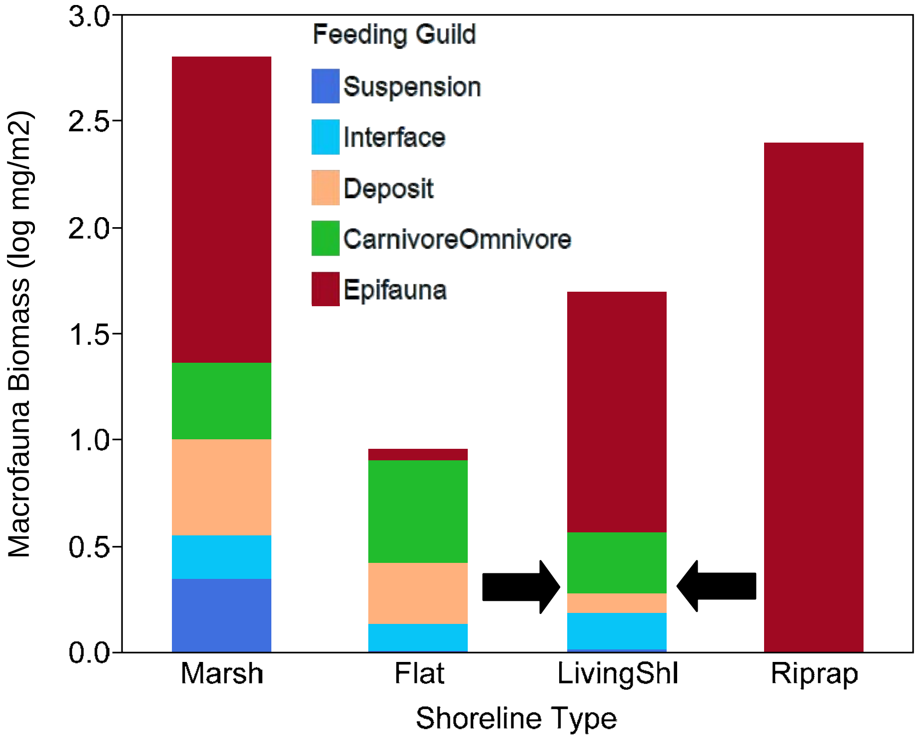


Figure 11. Macrobenthos community composition (2010-11) on the basis of individual feeding strategies, which are indicative of ecosystem service provision (e.g., epifauna = filter feeders that perform water filtration and can enhance water clarity).

3. What are the ecological tradeoffs of converting existing subtidal habitat to vegetated marsh-sill habitat?

All of the subtidal habitats (marsh, flat, riprap) consistently had greater biomass of suspension/interface and deposit feeders than the intertidal living shoreline (X^2=507.8, X^2=83.3, p <0.0001) (**Fig. 12**). Replacing **shallow subtidal** with **marsh-sill intertidal** may reduce infauna abundance, biomass and diversity as well as change the community structure. There is likely a loss of infauna suspension/interface and deposit feeders as a result of habitat conversion, with a gain in filter feeding epifauna that may offset some of the loss of infaunal filtration capacity, but not the loss of sediment mixing services ascribed to deposit feeders. In areas where shallow subtidal habitat is limited, the potential adverse effect on ecosystem services 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 is recommended to mitigate any potential effects on infauna. support higher tropic levels and maintain sediment stability. The unvegetated intertidal and shallow subtidal also provide refuge and feeding habitat for juvenile fish and invertebrates.

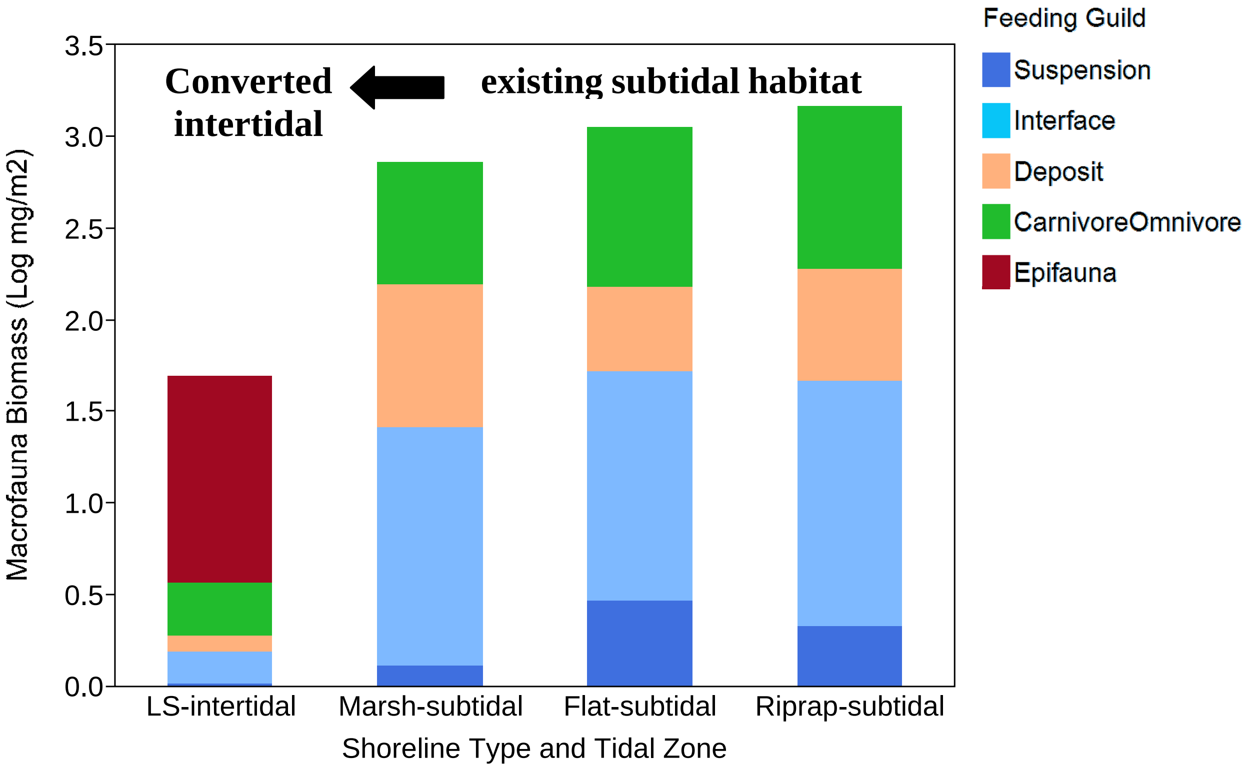


Figure 12. Comparison of macrobenthos composition (2010-11) at existing subtidal habitat that may be converted to the resulting converted habitat (i.e., intertidal vegetated marsh (LS-intertidal)).

4. Are macrobenthos communities in the subtidal habitats offshore of living shorelines similar to those offshore of natural shorelines?

Yes, There were generally no significant differences in infauna abundance, biomass or diversity in the subtidal adjacent to living shorelines in relation to natural wetlands (**Fig. 13**). One exception was that biomass associated with flats was greater than those near sills in 2010; however, biomass was similar between these shoreline types in 2011. The placement of living shorelines does not appear to adversely affect macrobenthos in adjacent shallow subtidal habitats.

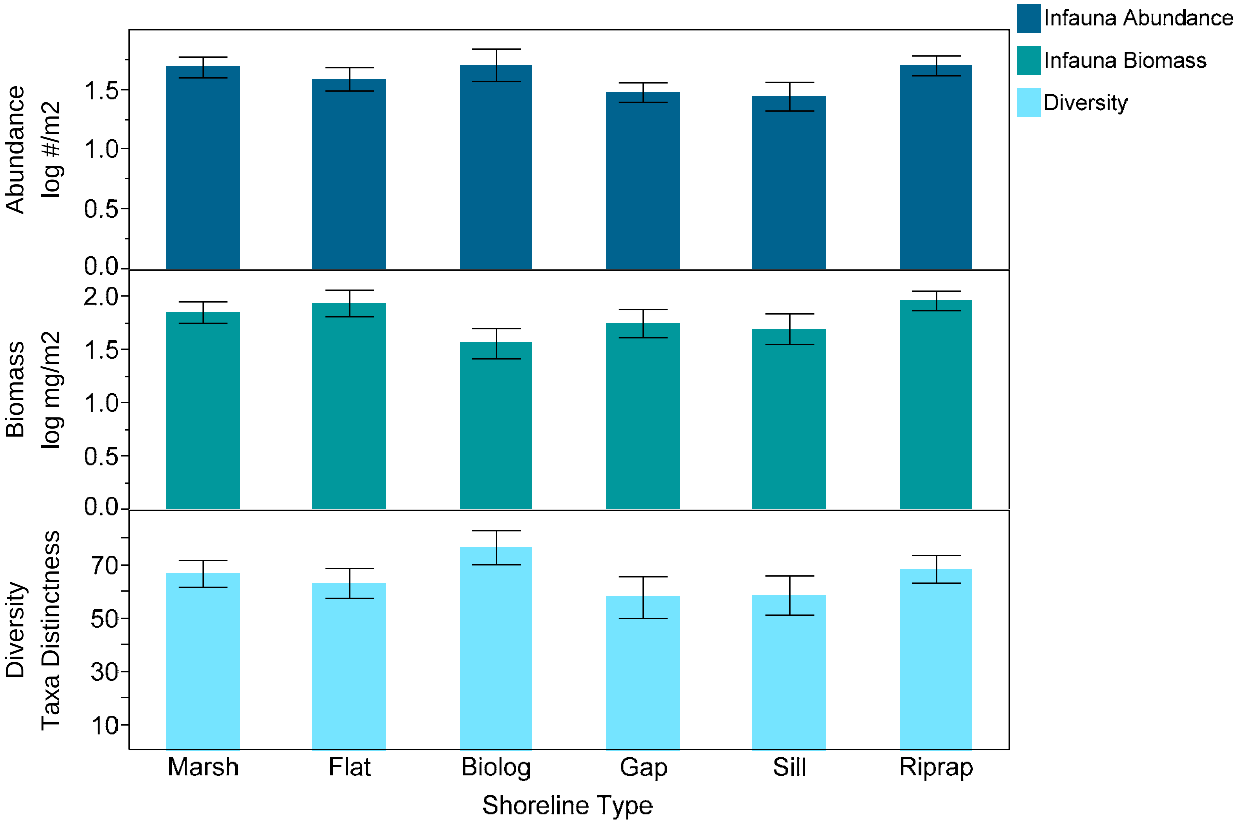
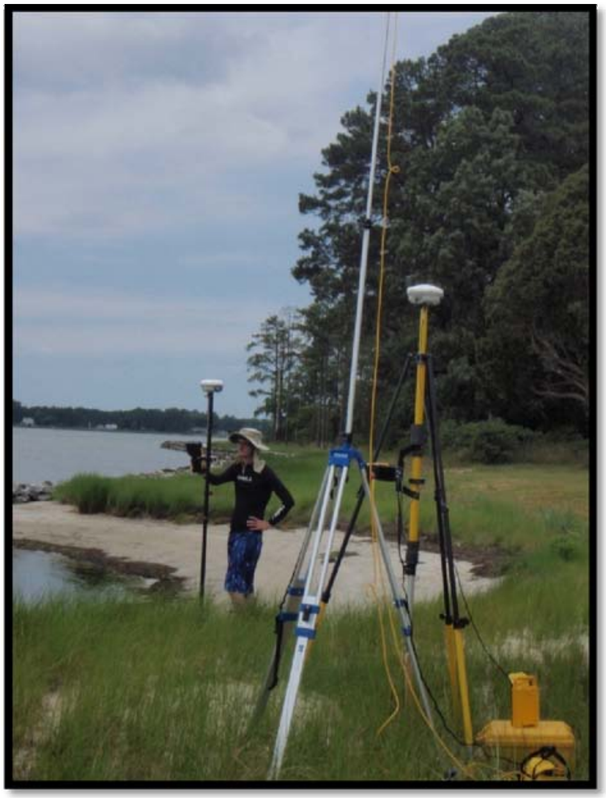


Figure 13. Macrobenthos abundance, biomass & diversity (±SE) in the subtidal were similar among shorelines.

5. Do marsh-sills provide comparable erosion protection to natural &/or riprap revetment shorelines?

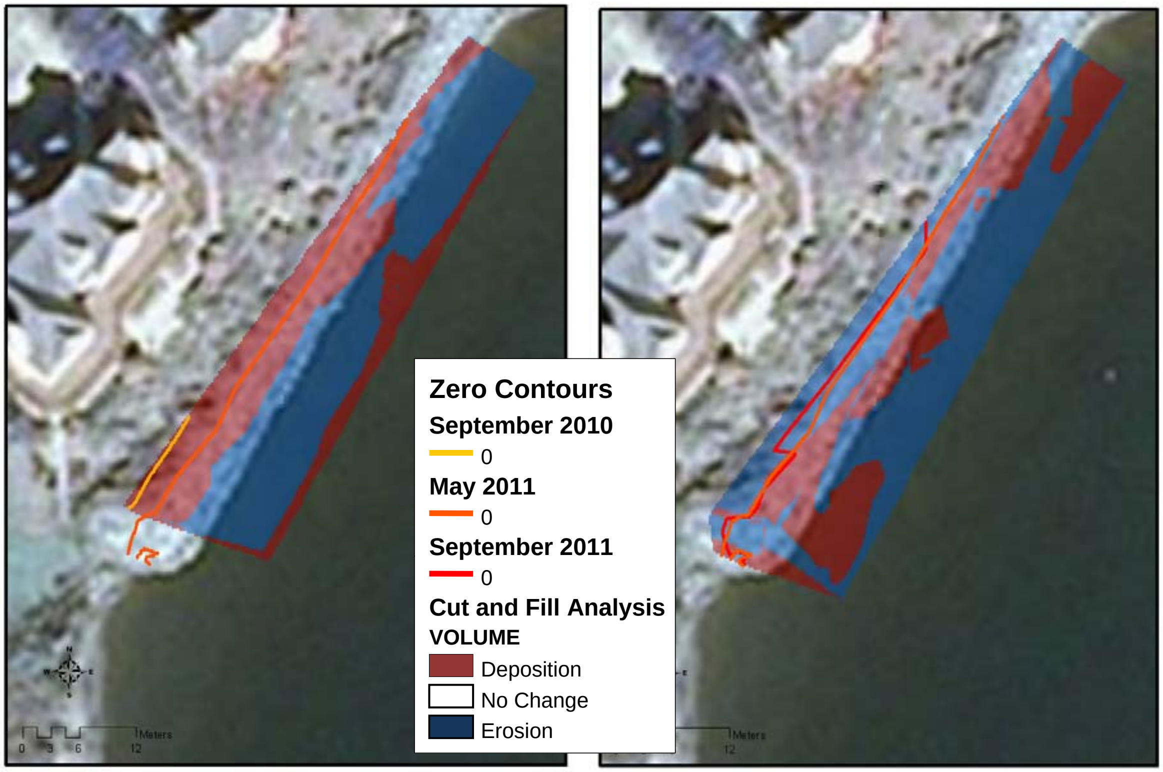
Yes.

Similar to the other shoreline types, the living shorelines maintained the location of the edge throughout time. They allowed for a certain amount of sediment movement both in front of and landward of the structure, showing accumulation of sediment on the marsh surface following the storm event, similar to the natural marshes. However, they appeared to also capture and retain sediment throughout the year, potentially increasing their stability and longevity relative to the natural marshes.



Net Gains & Loss of Sediment

South River Riprap Site

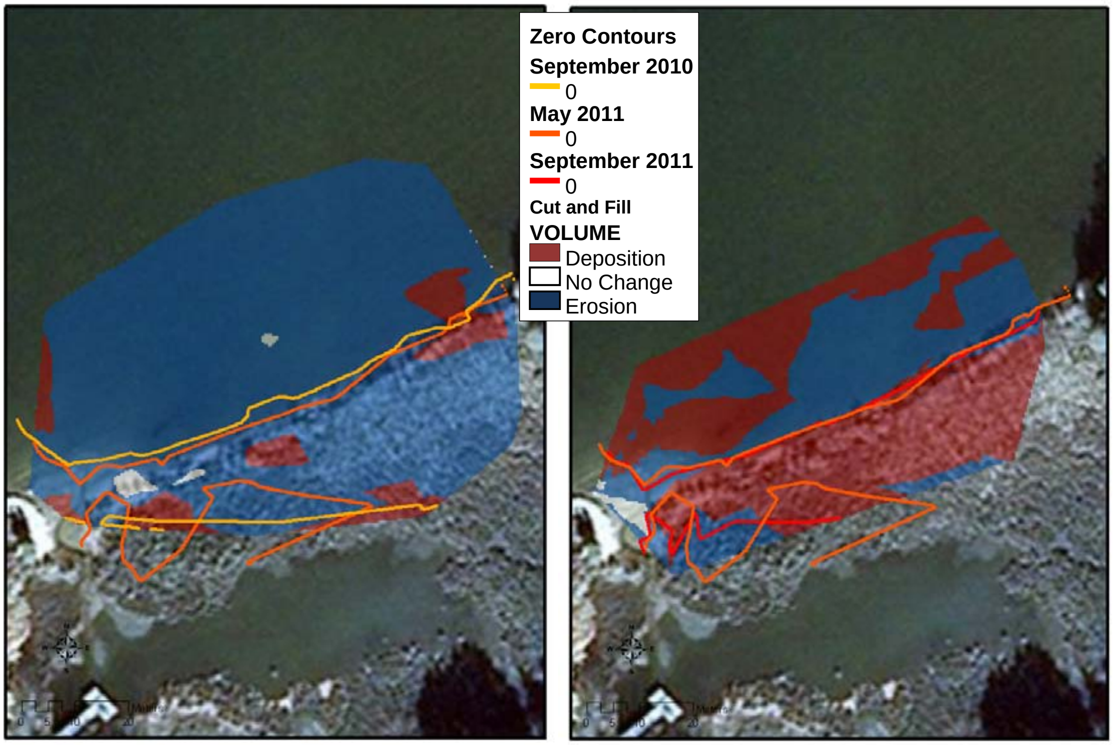

September 2010-May2011

May 2011-September 2011

The two rasters in the figure above represent volumetric changes in sediment between two surveys. The blue areas represent sediment lost and the red areas represent sediment gained. The method of raster creation resulted in a few artifacts, interpolation in the area above the zero contour line and some small edge effects. These result in a slight overestimation of deposition in the first figure and a slight overestimation of erosion in the second figure. Because overall volume change in the first time frame was a net gain and in the second time frame was a net loss, it is likely that this shoreline actually saw very little net change over the entire time period surveyed. The data do suggest there is some off-shore movement of sand in this area that may represent longshore sediment transport (i.e., a continual gain & loss of sediment moving along the shoreline) or a static sand supply which is moved and re-sorted through wave activities.

The zero contour line from September 2010 could not be completely projected along the shoreline due to a lack of data. However, May and September lines are very similar, which would be expected with a fully hardened shoreline.

South River Marsh Site



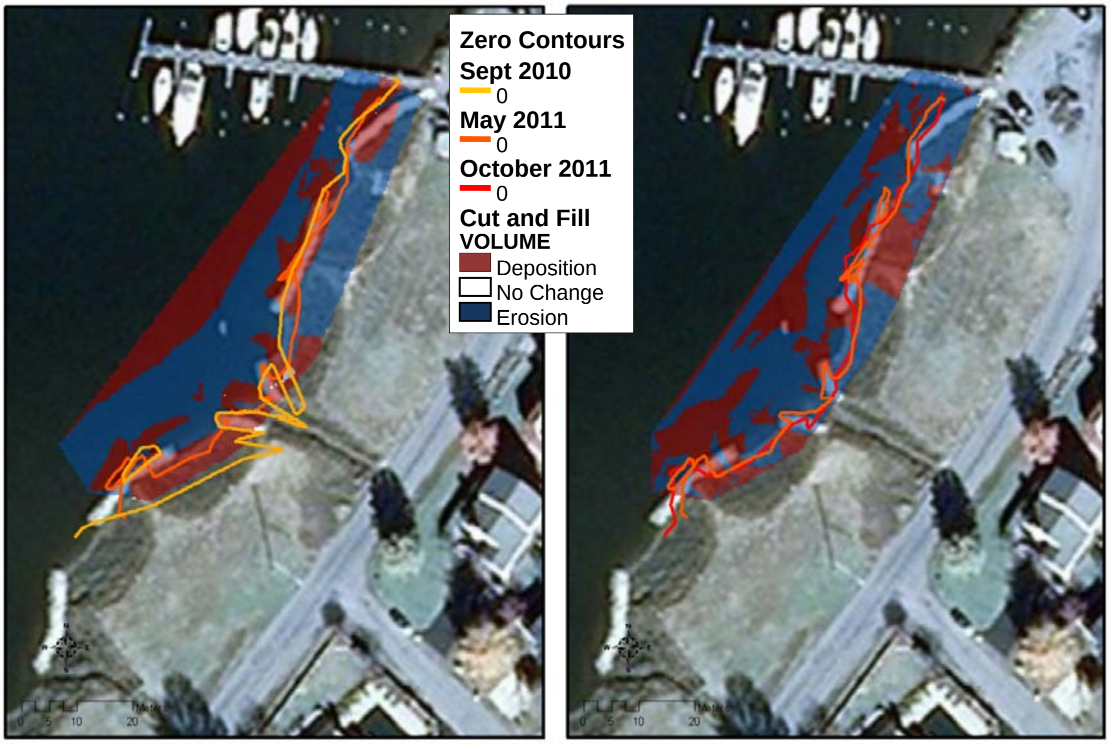
September 2010-May2011

May 2011-September 2011

The two rasters in the figure above represent volumetric changes in sediment between two surveys. The blue areas represent sediment lost and the red areas represent sediment gained. The method of raster creation resulted in a few artifacts including potentially some small edge effects on the western edge. However, in these rasters they appear small enough to minimally affect the overall results. In the first time frame, both the marsh surface and the offshore area appear to be predominately eroding. However, in the second time frame, the marsh surface appears to have gained some deposited sediment and there are areas of offshore gain as well. These suggest that the storm event may have brought sediment into an area which is typically eroding. Despite the deposition in the second time frame, the overall pattern at this location is net loss.

There are two zero contour lines shown for each time frame because the marsh surface declines into the marsh pond on the landward side. Similar to the results from the cut-and-fill analysis, the contour lines suggests shoreline erosion during the first time frame and little to or no shoreline migration in the second time frame. This may be due to sediment deposition during the storm event.

South River Living Shoreline Site

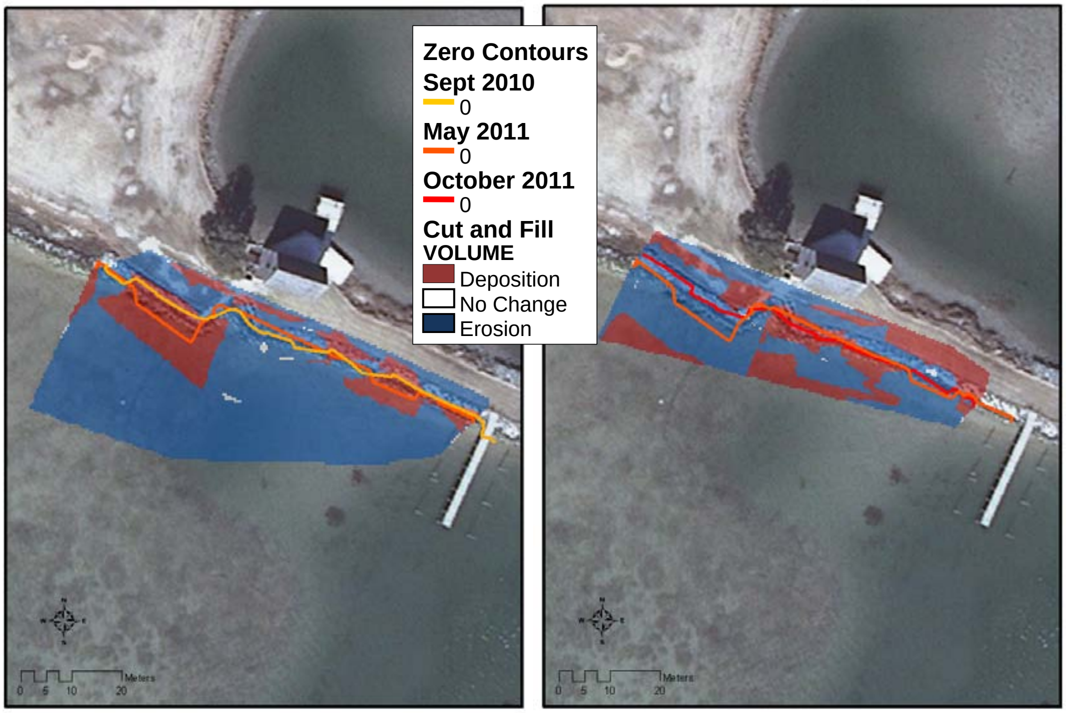

September 2010-May2011

May 2011-September 2011

The two rasters in the figure above represent volumetric changes in sediment between two surveys. The blue areas represent sediment lost and the red areas represent sediment gained. The method of raster creation resulted in a few artifacts including some small edge effects. However, in these rasters they appear small enough to minimally affect the overall results. In both time frames, there are areas of net gain and net loss, although in the first time frame the overall net movement is loss and while in the second time frame it is gain. There are no obvious patterns in the offshore sediment gain/loss, however there does appear to be a pattern of sediment gain immediately landward and seaward of the sills. These suggest that the sills are working to accrete sand on a shoreline that was previously eroding. The result is very little net change in elevations over the entire sampling period.

The zero contour lines in the first time frame show the contour moving offshore over time on the southern end of the sill. This movement is supported by the cut-and-fill analysis and likely shows accretion of sediment landward of the last surveyed sill. The zero contour lines in the second time frame are very similar, suggesting 1) that the shoreline is fairly stable and 2) that the sills are capable of holding captured sand during storm events.

East River Riprap Site


September 2010-May2011

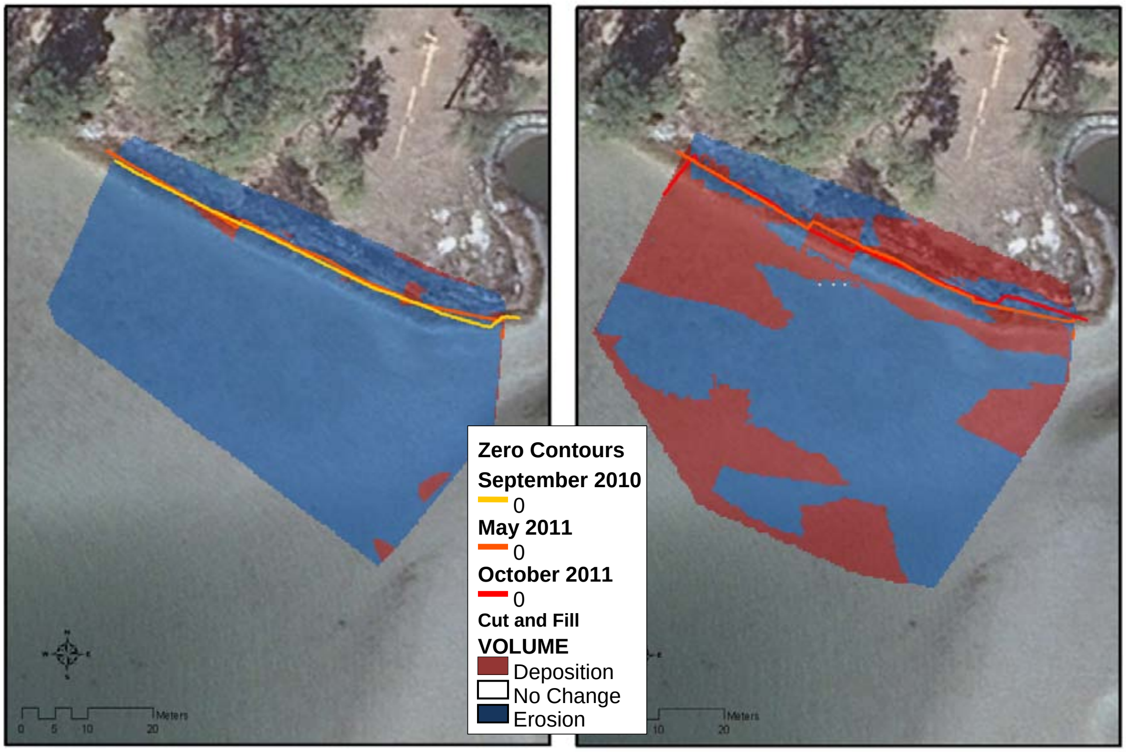
May 2011-October 2011

The two rasters in the figure above represent volumetric changes in sediment between two surveys. The blue areas represent sediment lost and the red areas represent sediment gained. The method of raster creation resulted in a few artifacts including some small edge effects. It is unclear in the second time frame how much of the deposition at the seaward edge of the survey area is real and how much is due to the edge effects. Therefore, deposition may overestimated in the total volume change. The coverage for the first time frame is larger than the second time frame because a low tide during the sampling events allowed surveying of a more extensive area. However, both time frames show consistent net erosion of the site. Overall, differences between the two time frames seem to represent more of a shift in sand accumulation patterns over time than a change in erosion processes.

The zero contour lines in both time frames show very little movement (as would be expected at a fully hardened shoreline) except at the western edge of the riprap where it ties into the adjacent living shoreline. The change at the western edge may be an artifact of the May 2011 sampling because the rock location did not change between sampling events and the September 2010 contour line matches more closely with the October 2011 contour line.

East Marsh Site

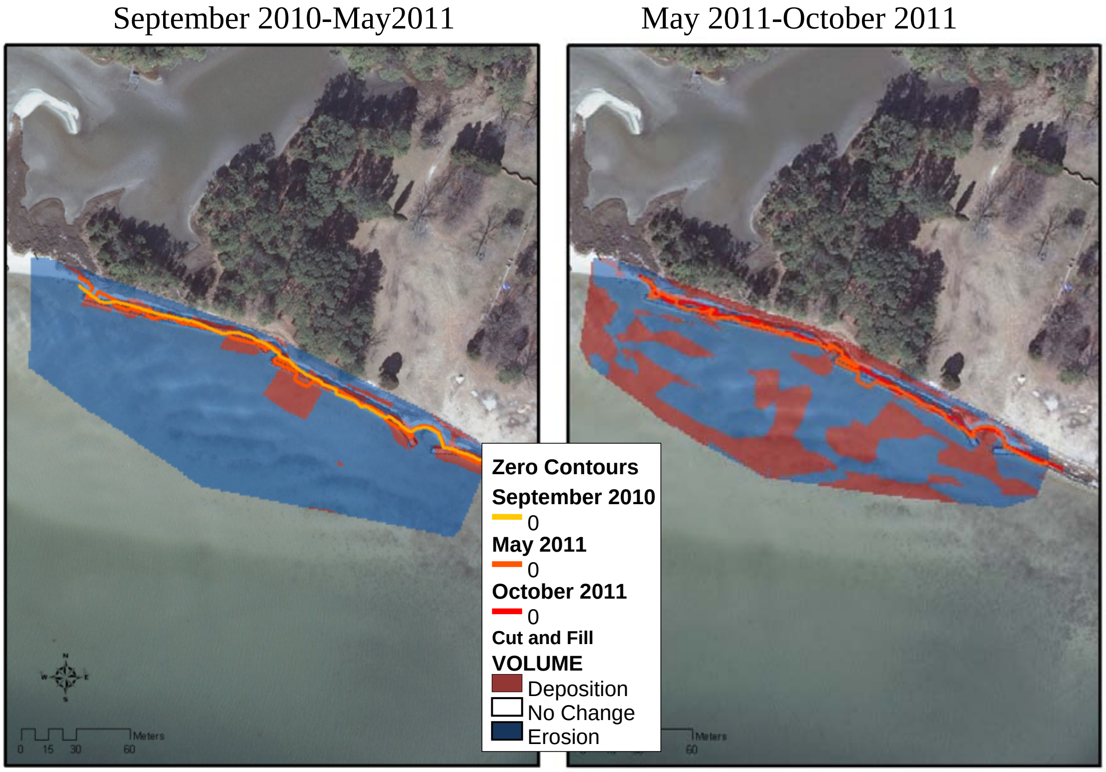
September 2010-May2011 May 2011-October 2011



The two rasters in the figure above represent volumetric changes in sediment between two surveys. The blue areas represent sediment lost and the red areas represent sediment gained. The method of raster creation resulted in a few artifacts including some small edge effects. However, in these rasters they appear small enough to minimally affect the overall results. In the first time frame, both the marsh surface and the offshore area appear to be predominately eroding. However, in the second time frame, the marsh surface appears to have gained some deposited sediment and there are areas of offshore gain as well. These suggest that the storm event may have brought sediment into an area which is typically eroding. Despite the deposition in the second time frame, the overall pattern at this location is net loss. These rasters, their patterns and changes in patterns are very similar to the South River marsh rasters.

The zero contour lines show little to no movement over the two time frames. This suggests that, on short time scales, the shoreline is fairly stable. However, the home owner at this property indicated that the shoreline has eroded significantly over the time period of his ownership which is consistent with the overall pattern of erosion at the site.

East Living Shoreline Site

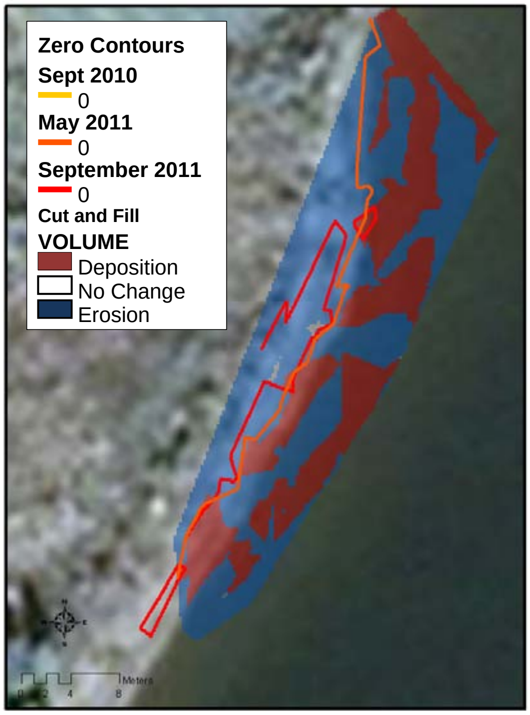


The two rasters in the figure above represent volumetric changes in sediment between two surveys. The blue areas represent sediment lost and the red areas represent sediment gained. The method of raster creation resulted in a few artifacts including some small edge effects. However, in these rasters they appear small enough to minimally affect the overall results. In the first time frame, both the marsh surface and the offshore area appear to be predominately eroding. However, in the second time frame, the marsh surface appears to have gained some deposited sediment and there are areas of offshore gain as well. These suggest that the storm event may have brought sediment into an area which is typically eroding. Sand waves in the aerial photo suggests that there is active reworking of the sediment along this shoreline, although it is impossible to tell if this represents longshore sediment transport or reworking of a static sediment supply. Despite the deposition in the second time frame, the overall pattern at this location is net loss. These rasters, their patterns and changes in patterns are very similar to both of the natural marsh rasters. Similar to the South River living shoreline, the sills appear to be “hot spots” for the collection of sediment on an otherwise eroding shoreline, and the sills seem capable of retaining the sediment during storm events.

The zero contour lines show little to no movement over the two time frames. This suggests that, on short time scales, the shoreline is fairly stable. It is not clear whether the sills are contributing towards the stability on the shoreline because the adjacent marsh (also an erosional system) had a stable shoreline over the sampling periods.

South River New Sill Site

May 2011-September 2011



The raster in the figure above represents volumetric changes in sediment between two surveys. The blue areas represent sediment lost and the red areas represent sediment gained. The method of raster creation resulted in a few artifacts including some small edge effects. However, in this raster they appear small enough to minimally affect the overall results. Similar to other living shoreline sites during this time frame, there is a mixture of erosion and deposition in the offshore area. However, unlike the other living shorelines, there is no indication on this site that the sill is capturing or retaining sediment on the landward side of the structure. In fact, all areas landward of the zero contour lines show erosion. This may be due to site specific characteristics or sill design. This sill is located in a higher energy setting and the northern end of this sill has a gap facing directly into the mouth of the river, subject to a fair amount of wave energy. This suggests that the energy climate and sill design may influence sediment retention efficiency and should be taken into consideration during the planning process.

The zero contour lines line up very well in some areas and not at all in other areas. The discrepancy between the lines is may be due to sampling issues; however, the pockets where the shoreline appears to have eroded may be reflecting an actual loss of sediment because the volumetric analysis is showing the same pattern.

Table 4. Volumetric and area changes in sediment between survey time periods. Negative changes indicate net gain and positive changes indicate net loss. Total volumetric change was standardized to the Area to allow relative comparisons between sites.

Site	Time frame	Δ Volume (m^3)	Area (m^2)	Δ Volume/Area (m) [in]
South River Riprap	Sep 2010-May 2011	-49	764	-0.06423 [-2.52]
	May 2011-Sept 2011	17	834	0.02009 [0.79]
South River Marsh	Sep 2010-May 2011	533	5085	0.10487 [4.13]
	May 2011-Sept 2011	-43	3420	-0.01269 [-0.50]
South River Living Shoreline	Sep 2010-May 2011	-42	2012	-0.02102 [-0.83]
	May 2011-Sept 2011	28	1807	0.01541 [0.61]
South River New Sill	May 2011-Sept 2011	24	538	0.004513 [0.18]
East River Riprap	Sep 2010-May 2011	104	2422	0.04307 [1.69]
	May 2011-Oct 2011	155	1420	0.10898 [4.29]
East River Marsh	Sep 2010-May 2011	252	2516	0.10025 [3.95]
	May 2011-Oct 2011	5	3239	0.00146 [0.06]
East River Living Shoreline	Sep 2010-May 2011	2438	16968	0.12328 [4.85]
	May 2011-Oct 2011	439	15405	0.02400 [0.94]

SUMMARY

Living shorelines provided shoreline stabilization, and may be following established created wetland trajectories (i.e., equivalence after 1-5 yrs for primary producers & 5-25 yrs for benthic infauna particularly subsurface deposit feeders (e.g., Craft et al. 2003)). Marsh plant communities were comparable to natural marshes in terms of density and plant height, which is representative of aboveground biomass. Following major storms Hurricane Irene and Tropical Storm Lee, elevation surveys of the marsh-sill living shorelines suggest that the shorelines were protected and the sills appear to be “hot spots” for the collection of sediment, and capable of retaining the sediment during storm events.

Other attributes of wetland structure, such as benthic infauna, develop more slowly than the plant community. Constructed salt marshes less than 20-25 years may have lower epifauna and infauna densities and fewer subsurface deposit feeders than in 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 age of the living shoreline should be considered during evaluation of ecosystem functioning. The surveyed living shorelines in this study were between 2 and 8 years of age and did not yet support equivalent infauna as natural marshes.

The placement of living shorelines involves the conversion of existing unvegetated intertidal and subtidal bottoms to a vegetated intertidal and/or rock sill. These existing shallow habitats 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 tropic levels (Middelberg et al. 2000) and maintain sediment stability (Madsen et al. 1993, Underwood and Patterson 1993). The unvegetated intertidal and shallow subtidal also provide refuge and feeding habitat for juvenile fish and invertebrates (Ruiz et al. 1993).

Evidence of ecological trade-offs occurring during habitat conversion include the enhancement of epifauna filter-feeders on sill structures with the reduction in infauna, particularly deposit-feeders. Therefore, there may be comparable water filtration capabilities in the living shorelines as natural marshes, but possibly a reduction in bioturbation by deposit feeders. When designing living shorelines 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 epifauna recruitment (e.g., oysters). There are numerous site dependent factors that will affect the recruitment and establishment of epifauna that should be considered to manage expectations of shoreline function. For example, oysters may not recruit to a given area due to unsuitable salinity or flow regime; therefore, 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 living shoreline designs (i.e., mussels & biologs). The continued exploration of living shoreline designs that incorporate a variety of biological components will allow for a robust array of alternatives that may more closely reflect natural conditions.

To identify structural and functional equivalence of living shoreline 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, fish). However, additional performance metrics are needed to evaluate marsh-sill as these hybrid designs marsh-sills are to some extent mimicking rocky intertidal habitats. 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 living shorelines do follow created marsh trajectories towards ecosystem equivalence.

ACKNOWLEDGEMENTS

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. Thanks to H. Berquist for assistance with shoreline change analyses. Special thanks to property owners Mr. Steve Hult and Ms. Teague for access to sites and informative conversations and to J. Davis and W. Priest for assistance with site selection.

LITERATURE CITED

Baird, D. and R.E. Ulanowicz. 1989. The seasonal dynamics of the Chesapeake Bay ecosystem. *Ecological Monographs* **59**:329-364.

Berlin, A. 2008. Foraging values of *Mulinia lateralis* and *Ischadium recurvum*: energetics effects of surf scoters wintering in the Chesapeake Bay. Ph.D. Dissertation. University of Maryland, College Park. 137 pp.

Bilkovic, D.M., and M. Roggero. 2011. Ecological and erosion protection functions of Chesapeake Bay living shorelines. Final Report to Chesapeake Bay Trust, National Oceanic and Atmospheric Administration (NOAA) Restoration Center, and the Maryland Department of the Environment. Center for Coastal Resources Management, Virginia Institute of Marine Science, Gloucester Point, VA. http://ccrm.vims.edu/publications/pubs/Bilkovic_CBTrust_FinalReport_13Dec2011.pdf

Bilkovic, D.M., and M. Roggero. 2008. Effects of coastal development on nearshore estuarine nekton communities. *Marine Ecology Progress Series* **358**:27-39.

Bilkovic, D.M., M. Roggero, C.H. Hershner, and K. Havens. 2006. Influence of land use on macrobenthic communities in nearshore estuarine habitats. *Estuaries and Coasts* **29**:1185-1195.

Bilkovic, D.M., C.H. Hershner, M.R. Berman, K.J. Havens and D.M. Stanhope. 2005. Evaluating Nearshore Communities as Indicators of Ecosystem Health. *In Estuarine Indicators*, S.A. Bortone (ed.). CRC Press, Inc. Boca Raton, Florida. pp 365-379.

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. Columbia: University of South Carolina Press pp. 245-266.

Buchanan, C., R.V. Lacouture, H.G. Marshall, M. Olson, and J. Johnson. 2005. Phytoplankton reference communities for Chesapeake Bay and its tidal tributaries. Estuaries 28:138–159.

Burke, David G., Evamaria W. Koch and J. Court Stevenson. 2005. *Assessment of Hybrid Type Shore Erosion Control Projects in Maryland's Chesapeake Bay – Phases I & II*. Final report for Chesapeake Bay Trust, Annapolis, Maryland.

Carroll, R.A. 2002. Nekton utilization of intertidal fringing salt marsh and revetment hardened shorelines. Doctoral dissertation. The College of William and Mary. 109 pp.

Chambers, R. M., L. A. Meyerson, and K. Saltonstall. 1999. Expansion of *Phragmites australis* into tidal wetlands of North America. *Aquatic Botany* **64**:261–274.

Clarke K.R. and R.M. Warwick. 1999. The taxonomic distinctness measure of biodiversity: weighting of step lengths between hierarchical levels. *Marine Ecology, Progress Series* **184**:21–29.

Cohen, R.R. H., P.V. Dresler, E.J.P. Phillips, and R.L. Cory. 1984. The effect of the Asiatic clam, *Corbicula fluminea*, on phytoplankton of the Potomac River, Maryland. *Limnology and Oceanography* **29**:170-180.

Craft, C., P Megonigal, S Broome, J Stevenson, R Freese, J Cornell and J Sacco. 2003. The pace of ecosystem development of constructed *Spartina alterniflora* salt marshes. *Ecological Applications* **13**:1417–1432.

Davis, J.L.D., R.L. Takacs and R. Schnabel, 2006. Evaluating ecological impacts of living shorelines and shoreline habitat elements: an example from the upper western Chesapeake Bay, *CRC Publ. No. 08-164, Chesapeake Bay*, in: S.Y. Erdle, J.L.D. Davis, K.G. Sellner, Editors, *Management, Policy, Science, and Engineering of Nonstructural Erosion Control in the Chesapeake Bay*, pp. 55–61.

Deegan, L., J. Bowen, D. Drake, J. Fleeger, C. Friedrichs, K. Galvan, J. Hobbie, C. Hopkins, D. Johnson, J. Johnson, L. LeMay, E. Miller, B. Peterson, C. Picard, S. Sheldon, M. Sutherland, J. Vallino and R. Warren. 2007. Susceptibility of salt marshes to nutrient enrichment and predator removal. *Ecological Applications* 17(5):S42-S63.

Diaz, R.J. and L.C. Schaffner. 1990. The Functional Role of Estuarine Benthos, p. 25-56. *In:* M. Haire and E.C. Krome (eds.), *Perspectives on the Chesapeake Bay, 1990. Advances in Estuarine Sciences.* United States Environmental Protection Agency, Gloucester Point, VA.

Folk, R.L. 1980, Petrology of sedimentary rocks: Hemphill Pub. Co., Austin, Texas.

Grant, W.D., L.F. Boyer, and L.P. Sanford. 1982. The effects of bioturbation on the initiation of motion of intertidal sands. *Journal of Marine Research* **40**:659-677.

Hughes, R.N. and R. Seed. 1981. Size selection of mussels by the blue crab *Callinectes sapidus*: energy maximizer or time minimizer? *Marine Ecology Progress Series* **6**:83–89

King, R.S., A.H. Hines, F.D. Craige, and S. Grap. 2005. Regional, watershed and local correlates of blue crab and bivalve abundances in subestuaries of Chesapeake Bay, USA. *Journal of Experimental Marine Biology and Ecology* **319**: 101-116.

King, R.S., W.V. Deluca, D.F. Whigham, and P.P. Marra. 2007. Threshold effects of coastal urbanization on *Phragmites australis* (common reed) abundance and foliar nitrogen in Chesapeake Bay. *Estuaries and Coasts* **30**:1–13.

Lacouture, R.V., J.M. Johnson, C. Buchanan and H.G. Marshall. 2006. Phytoplankton index of biotic integrity for Chesapeake Bay and its tidal tributaries. *Estuaries and Coasts* **29**:598–616

Levin, L.A., D. Talley, and G. Thayer. 1996. Succession of macrobenthos in a created salt marsh. *Marine Ecology Progress Series* **141**:67–82.

MacIntyre, H.L., R.J. Geider, and D.C. Miller. 1996. Microphytobenthos: The ecological role of the “secret garden” of unvegetated, shallow-water marine habitats. I. Distribution, abundance and primary production. *Estuaries and Coasts* **19**:186-201.

Madsen, K.N., P. Nilsson, and K. Sundback. 1993. The influence of benthic microalgae on the stability of a subtidal sediment. *Journal of Experimental Marine Biology and Ecology* **170**:159-177.

Mann, R., M. Southworth, J. M. Harding and J. Wesson. 2009. Population studies of the native oyster, *Crassostrea virginica* (Gmelin), in the James River, Virginia, USA. *J. Shellfish Res.* **28**:193–220.

Middelburg, J.L., H.T.S. Boschker, P.M.J. Herman, T. Moens, and C.H.R. Heip. 2000. The fate of intertidal microphytobenthos carbon: An in situ 13C-labelling study. *Limnology and Oceanography* **45**:1224-1234.

Miller, D.C., R.J. Geider, and H.L MacIntyre. 1996. Microphytobenthos: The ecological role of the “secret garden” of unvegetated, shallow-water marine habitats. II. Role in sediment stability and shallowwater food webs. *Estuaries and Coasts* **19**:202-212.

Moy, L.D., and L.A. Levin. 1991. Are *Spartina* marshes a replaceable resource? A functional approach to evaluation of marsh creation efforts. *Estuaries and Coasts* **14**:1–16.

Neubauer, M.T. 2000. Demographics, production, and benthic-pelagic coupling by the suspension feeding polychaete *Chaetopterus pergamentaceus* in the Lower Chesapeake Bay. Ph.D. Dissertation. The College of William and Mary, Williamsburg, VA.

Newell, R. 1988. Ecological changes in Chesapeake Bay; are they a result of over-harvesting the American oyster Crassostrea virginica? Understanding the Estuary; Advances in Chesapeake Bay Research. Baltimore: *Chesapeake Research Consortium Publication* 29.

Rhoads, D.C. and D.K. Young. 1970. The influence of deposit-feeding organisms on sediment stability and community trophic structure. *Journal of Marine Research* **28**:150-178.

Ruiz, G.M., A.H. Hines, and M.H. Posey. 1993. Shallow water as a refuge habitat for fish and crustaceans in non-vegetated estuaries: An example from Chesapeake Bay. *Marine Ecology Progress Series* **99**:1-16.

Sacco, J.N., E.D. Seneca, and T. Wentworth. 1994. Infaunal community development of artificially established salt marshes in North Carolina. *Estuaries and Coasts* **17**:489–500.

Scatolini, S.R., and J.B. Zedler. 1996. Epibenthic invertebrates of natural and constructed marshes of San Diego Bay. Wetlands **16**:24–37.

Seitz, R.D., R.N. Lipcius, N.H. Olmstead, M.S. Seebo, and D.M. Lambert. 2006. Influence of shallow-water habitats and shoreline development on abundance, biomass, and diversity of benthic prey and predators in Chesapeake Bay. *Marine Ecology Progress Series* **326**:11-27.

Snelgrove, P. 1998. The biodiversity of macrofaunal organisms in marine sediments. *Biodiversity and Conservation* **7**:1123-1132.

Snelgrove, P.V.R. and C.A. Butman. 1994. Animal-sediment relationships revisited: cause versus effect. *Oceanography and Marine Biology: an Annual Review* **32**:111-177.

Tisdale A.L, W.L. Nelson, and J.D. Beaton. 1985. Soil Fertility and Fertilizers. Macmillan Pub. Co., New York.

Tobias, C., S. Mako, I. Anderson, E. Canuel and J. Harvey. 2001. Tracking the fate of a high concentration groundwater nitrate plume through a fringing marsh: A combined groundwater tracer and *in situ* isotope enrichment study. *Limnology and Oceanography* 46(8):1977-1989.

Tyler, A.C., K.J. McGlathery, and I. Andersen. 2003. Benthic algae control sediment-water column fluxes of organic and inorganic nitrogen compounds in a temperate lagoon. *Limnology and Oceanography* 48, 2125-2137.

Underwood, G.J.C., and D.M. Patterson. 1993. Seasonal changes in diatom biomass, sediment stability and biogenic stabilization in the Severn Estuary. *Journal of the Marine Biological Association of the United Kingdom* **76**:431-450.

Whitlatch, R.B. 1980. Patterns of resource utilization and coexistence in marine intertidal deposit- feeding communities. *Journal of Marine Research* **38**:743-765.

Appendix I. List of observed macrobenthos species & associated feeding guilds, 2010-2011

Species	Family	Order	Class	Phylum	Feeding Guild
Gemma gemma	Veneridae	Veneroida	Bivalvia	Mollusca	Suspension
Tagelus plebeius	Solecurtidae	Veneroida	Bivalvia	Mollusca	Suspension
Tagelus divisus	Solecurtidae	Veneroida	Bivalvia	Mollusca	Suspension
Macoma balthica	Tellinidae	Veneroida	Bivalvia	Mollusca	Interface
Tellina agilis	Tellinidae	Veneroida	Bivalvia	Mollusca	Interface
Rangia cuneata	Mactridae	Veneroida	Bivalvia	Mollusca	Suspension
Mulinia lateralis	Mactridae	Veneroida	Bivalvia	Mollusca	Suspension
Geukensia demissa	Mytilidae	Mytiloida	Bivalvia	Mollusca	Suspension
Mytilopsis leucophaeata	Dreissenidae	Mytiloida	Bivalvia	Mollusca	Suspension
Heteromastus filiformis	Capitellidae	Capitellida	Polychaeta	Annelida	Subsurface deposit feeder
Clymenella torquata	Maldanidae	Capitellida	Polychaeta	Annelida	Subsurface deposit feeder
Capitellidae spp	Capitellidae	Capitellida	Polychaeta	Annelida	Subsurface deposit feeder
Neanthes succinea	Nereididae	Phyllodocida	Polychaeta	Annelida	Carnivore/Omnivore
Eteone heteropoda	Phyllodocidae	Phyllodocida	Polychaeta	Annelida	Carnivore/Omnivore
Glycera americana	Glyceridae	Phyllodocida	Polychaeta	Annelida	Carnivore/Omnivore
Glycera dibranchiata	Glyceridae	Phyllodocida	Polychaeta	Annelida	Carnivore/Omnivore
Glycera capitata	Glyceridae	Phyllodocida	Polychaeta	Annelida	Carnivore/Omnivore
Leitoscoloplos fragilis	Orbiniidae	Orbiniida	Polychaeta	Annelida	Subsurface deposit feeder
Amphitrite ornata	Terebellidae	Terebellida	Polychaeta	Annelida	Interface
Loimia medusa	Terebellidae	Terebellida	Polychaeta	Annelida	Interface
Melinna maculata	Ampharetidae	Terebellida	Polychaeta	Annelida	Interface
Spiochaetopterus oculatus	Chaetopteridae	Spionida	Polychaeta	Annelida	Interface
Marenzelleria viridis	Spionidae	Spionida	Polychaeta	Annelida	Interface
streblospio benedicti	Spionidae	Spionida	Polychaeta	Annelida	Interface
Spionidae spp	Spionidae	Spionida	Polychaeta	Annelida	Interface
Polydora cornuta	Spionidae	Spionida	Polychaeta	Annelida	Interface
Spiophanes bombyx	Spionidae	Spionida	Polychaeta	Annelida	Interface
Spio setosa	Spionidae	Spionida	Polychaeta	Annelida	Interface
Lumbrineridae spp	Lumbrineridae	Aciculata	Polychaeta	Annelida	Carnivore/Omnivore
Glycinde solitaria	Goniadidae	Aciculata	Polychaeta	Annelida	Carnivore/Omnivore
Phoronids spp	Phoronidae	Phoronidaea	Phoronida	Phoronida	Suspension
Oligochaeta spp	Oligocheatae	Oligocheata	Clitellata	Annelida	Subsurface deposit feeder
Haustonidae spp	Haustonidae	Ampipoda	Malacostraca	Arthropoda	Subsurface deposit feeder
Corophium lacustre	Corophiidae	Ampipoda	Malacostraca	Arthropoda	Interface
Listriella clymenella	Liljeborgiidae	Ampipoda	Malacostraca	Arthropoda	Carnivore/Omnivore
Gammarus spp	Gammaridae	Ampipoda	Malacostraca	Arthropoda	Carnivore/Omnivore
Hargeria rapax	Leptocheliidae	Tanaidacea	Malacostraca	Arthropoda	Interface
Cyathura polita	Anthuridae	Isopoda	Malacostraca	Arthropoda	Carnivore/Omnivore
Ericsonella attenuata	Idoteidae	Isopoda	Malacostraca	Arthropoda	Carnivore/Omnivore
Edotea triloba	Idoteidae	Isopoda	Malacostraca	Arthropoda	Carnivore/Omnivore
Chiridotea almyra	Chaetiliidae	Isopoda	Malacostraca	Arthropoda	Carnivore/Omnivore
Collembola spp	Collembolae	Collembola	Insecta	Arthropoda	Carnivore/Omnivore
Chironomid larvae	Chironomidae	Diptera	Insecta	Arthropoda	Carnivore/Omnivore
Rhithropanopeus harrisii	Xanthidae	Decapoda	Crustacea	Arthropoda	Carnivore/Omnivore
Cumacean spp	Cumacidae	**Cumacea**	Crustacea	Arthropoda	Carnivore/Omnivore