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## The Consequences of Shoreline Development for Near-Shore Communities in Chesapeake Bay, USA: A Before-After Control-Impact Study

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**The consequences of shoreline development for near-shore communities in  
Chesapeake Bay, USA: a before-after control-impact study**

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A Thesis

Presented to

The Faculty of the School of Marine Science

The College of William and Mary in Virginia

In partial fulfillment  
of the Requirements for the Degree of  
Master of Science

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by

Theresa M. Davenport

2012

APPROVAL PAGE

This Thesis is submitted in partial fulfillment of  
the requirements for the degree of

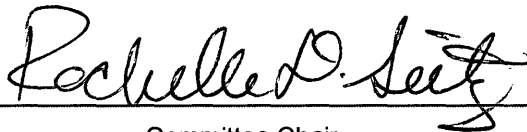
Master of Science



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Theresa Marie Davenport

Approved by the Committee, December 2012



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Committee Chair  
Rochelle D. Seitz, Ph.D.



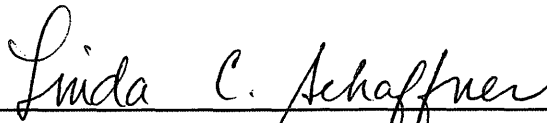
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Linda C. Schaffner, Ph.D.

## **DEDICATION**

This thesis is dedicated to the memory of Margaret J. (1930-2002) and Cletus D. Davenport (1923-2005). Their wisdom, generosity, and patience through countless adventures to the great outdoors have sparked my interest in and respect for the natural world, especially all things aquatic.

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## ABSTRACT

Hardened shorelines and their construction introduce stressors to a system by altering near-shore habitats. They can reduce biodiversity and abundance of benthic infauna and marsh-edge nekton and epifauna. In this study, I investigated the impacts of shoreline development on near-shore communities using a temporal and spatial approach, by use of a before-after control-impact (BACI) study design at four sub-estuaries within Chesapeake Bay that represent three different types of shoreline change. The BACI study was used to examine infaunal density, biomass, and diversity for two size classes of infauna (3-mm: larger species and adults, 500- $\mu$ m: smaller species and juveniles), as well as abundance of blue crabs and abundance and diversity of near-shore fishes before and after shorelines were modified.

Data were analyzed with Akaike's Information Criterion to compare candidate sets of linear models that contained year, shoreline treatment, sediment grain size, and salinity as predictors. In response to shoreline modification, infaunal density and biomass increased at sites that were newly developed (Timberneck, Dandy, and Holly Cove), but decreased at the site that changed from bulkhead to living shoreline (Windy Hill). In addition, infaunal diversity decreased at Timberneck and Windy Hill. Blue crab abundance increased at Timberneck, remained constant at Dandy, and decreased at Holly Cove. Blue crab abundance decreased at Windy Hill, though this may not be a shoreline modification response, as crabs concomitantly decreased at control shorelines. Fish abundance and diversity showed no distinct shoreline response at any site, which may reflect their transient nature.

At Timberneck, infaunal responses to shoreline modification were mainly driven by changes in habitat. At Dandy, and Holly Cove, infaunal responses to shoreline modification were mainly driven by opportunistic species. At Windy Hill, infaunal responses were driven by a more uniform reduction in all species after the conversion. Sediment grain size was an important predictor of infaunal response variables at Timberneck and Holly Cove, and also changed concomitantly with shoreline modification at Timberneck. Salinity was an infrequent predictor of infaunal responses, though it did vary between years.

The importance of opportunistic species and changes in sediment grain size one year after shoreline modification emphasizes the need to monitor shoreline modifications as ecological disturbances and changes in habitat, and the need to consider the role of shoreline modification in ecological succession rather than compare shorelines as static habitats. Lengths of shorelines that were modified were generally short, except at Windy Hill. Negative impacts of shoreline modification at Windy Hill likely reflect a larger disturbance than other sites, and a longer time needed to see positive improvements expected with a living shoreline. Fish species are more motile in nature, may be faster positive responders to positive changes in shoreline condition than infaunal benthos, and provided a first look at the positive effects of living shorelines at Windy Hill.

The consequences of shoreline development for near-shore communities in Chesapeake Bay, USA: a before-after control-impact study

## INTRODUCTION

### *Ecosystem Function and Shoreline Development in Mid-Atlantic Estuaries*

Within the Mid-Atlantic United States, estuarine subtidal benthic fauna provide many ecosystem functions such as secondary production and trophic transfer (Valiela 1995, Seitz et al. 2003, 2005), benthic-pelagic coupling (Caliman et al. 2007), and cycling and storing toxicants (Diaz and Schaffner 1990). Infauna in Chesapeake Bay are responsible for production of about 194,000 metric tons C yr<sup>-1</sup>, supporting 27,500 mt of fisheries yield (Diaz and Schaffner 1990). Benthic infauna serve as critical links (benthic-pelagic couplers) between sources of organic matter in estuaries, consuming carbon from phytoplankton, benthic microalgae and macroalgae, and detritus, and providing food for economically and ecologically important finfish and crustaceans, such as the blue crab *Callinectes sapidus* (Seitz et al. 2001, 2003, Caliman et al 2007, Gillett and Schaffner 2009). About 50 % of fish production in Chesapeake Bay is linked to the benthic food web (Baird and Ulanowicz 1989), and benthic-feeding (demersal) nekton landings yielded an average of \$39.8 million of revenue between 1998-2002 (Gillett and Schaffner 2009).

Benthic communities affect water and sediment quality in estuaries. Filter feeding organisms in the benthos remove particles from the water column, leading to faster deposition than natural sinking can provide (Graf and Rosenberg 1997), resulting in enhanced water clarity and SAV coverage (Gillett and Schaffner 2009). Biodeposition by filter feeders may transfer water-column production to the sediment bed, where benthic, not pelagic, processes determine transport, transformation, and fates of sediments (Cohen et al. 1984). Bioturbation, or biological structuring and reworking of

the sediment surface by benthic organisms, has major effects on nutrient and contaminant cycling and fate (Diaz and Schaffner 1990), enhancing breakdown of organic matter and contaminants.

Tidal marshes in Mid-Atlantic estuaries such as Chesapeake Bay are important in sustaining high levels of primary production, contributing to detritus production and availability, supporting wildlife and waterfowl, buffering shorelines to prevent erosion, and maintaining water quality (Bertness 2007, Perry and Atkinson 2009). Primary productivity in tidal wetlands of the Chesapeake can reach 4 metric tons  $\text{ha}^{-1} \text{yr}^{-1}$ , leading to high levels of detritus production and providing the base of a major source of marine trophic transfer to crabs, other shellfish, and finfish among others (Lipcius et al. 2005, Perry and Atkinson 2009). Additionally, tidal marshes provide spawning and nursery habitat (Minello et al. 2003), with an estimated 95% of Virginia's annual fish harvest from tidal waters depending on wetlands to some degree (Wass and Wright 1969). Blue crabs, oysters, clams, striped bass, spot, croaker, and menhaden represent some of the important wetland-dependent fisheries in Chesapeake Bay (Perry and Atkinson 2009).

For blue crabs, tidal marshes in the York River serve a nursery role that is nearly as important as that of seagrass, especially in upriver, non-SAV-dominated sites (Lipcius et al. 2005). Tidal marshes are also responsible for maintaining and improving water quality, as they trap sediment and potentially pollution from upland runoff and the water column, and reduce siltation of shellfish beds, SAV areas, and navigation channels (Perry and Atkinson 2009). In addition to their biological functions, tidal marshes dissipate incoming wave energy, providing a natural buffer against shoreline erosion and acting as

the least erodible natural shoreline (Rosen 1980); over 50% of wave energy is dissipated in the first 2.5 m of a *Spartina alterniflora* marsh (Knutson et al. 1982).

Shoreline erosion is a natural process in estuaries, but can be exacerbated by human activity. Roughly one third of shorelines in Chesapeake Bay are classified as eroding, with some areas losing as much as 20 – 40 cm of shoreline per year (Chesapeake Bay Program 2005). Meanwhile, coastal landscapes, which surround most major worldwide cities and 53% of the U.S. population, are being transformed rapidly to meet the demands of commercial, residential, and tourist activities (Bulleri and Chapman 2010) including shipping, fisheries, recreation, and transportation among other uses. The Chesapeake Bay watershed's human population is currently home to over 17.5 million people, rapidly growing by about 157,000 people per year, and projected to grow to 20 million by 2030 (Chesapeake Bay Program 2012), and with population growth comes increased environmental pressure from upland and shoreline development as a larger population utilizes coastal resources. Over half (60%) of Virginia's population lives on the coastal plain which makes up just under 25% of the state's land mass (Rappaport and Sachs 2003, Perry and Atkinson 2009), and ~70 to 90% of Virginia and Maryland's populations live in coastal counties (Crossett et al. 2004), putting pressure on coastal systems and salt marshes.

In Chesapeake Bay, some sub-watersheds' coastlines are over 50% "hardened" to protect against damage to infrastructure, replacing natural shoreline habitats with artificial hard surfaces, such as bulkheads and riprap revetments (Berman et al. 2000). Since the 1970s, hundred of miles of shoreline in Virginia and Maryland have been armored (Titus 1998, Barnard 2004). Even as artificial substrates are becoming



increasingly ubiquitous in intertidal and shallow subtidal habitats, related ecological issues have received little attention (Halpern et al. 2008), presumably due to a lack of information (Bulleri and Chapman 2010).

### ***Shoreline Development as a Stressor***

Shoreline development can be operationally defined as waterfront development with anthropogenic structure (Silliman and Bertness 2004, Seitz et al. 2006, Bulleri and Chapman 2010). Placement of anthropogenic structures is an environmental disturbance that introduces stressors to a system, such as alteration of the natural shape, near-shore wave dynamics, and shallow water habitat of the shoreline, and habitat fragmentation (Bilkovic and Roggero 2008, Bulleri and Chapman 2010).

Coastal development with hard structures, such as boat docks, bulkheads, and riprap represent a substantial disturbance to near-shore environments and reduction in their performance of ecosystem services. In Waquoit Bay, Massachusetts, eelgrass decreased in density under and directly adjacent to boat docks, as light availability decreased under the docks, shifting in position throughout the day with the angle of sun (Burdick and Short 1999). This may also translate to reduced benthic microalgae and/or phytoplankton reserves for benthic infauna, as well as a reduced ability for seagrass beds to perform sediment stabilization and provide habitat (Burdick and Short 1999). In Mobile Bay, Alabama, bulkhead shorelines have led to a complete loss of adjacent intertidal habitat, with length of impacted shoreline increasing with population growth from the mid-1950s to mid-1990s to encompass 30% of the 100-mile long shoreline at that time (Douglass and Pickel 1999).

Shoreline structures alter the physical characteristics of the environment and affect adjacent infaunal communities. Bulkheads typically increase wave reflection and sediment scour, causing a decrease in the width of the near-shore environment and increasing water depth, and often affect adjacent beaches (Douglass and Pickel 1999). Depth and sediment types at bulkhead shorelines are significantly deeper (Jivoff 2005) and contain coarser sediments such as gravels (Ahn and Choi 1998, Jivoff 2005). Thus, bulkheads contribute to the narrowing and eventual loss of intertidal beaches in front of them by contributing to sediment loss, at which point tidal differences are only seen as water height against the vertical bulkhead (Douglass and Pickel 1999), and sometimes lead to their own failure if sediment scour continues unchecked. Riprap revetments are typically sloped structures parallel to the shoreline and are constructed to protect the shoreline from erosion and absorb wave energy (Burcharth and Hughes 2002, Bendell 2006). Wave energy absorption at riprap shorelines is improved over bulkheads, but riprap also deepens water and concentrates turbulence, alters the physical structure of the shoreline, alters patterns of sedimentation and erosion, and replaces vegetation with hard structures (Burcharth and Hughes 2002, Bendell 2006).

While bulkhead and riprap can provide effective erosion control, they alter natural habitats in ways that reduce their ability to perform important ecosystem services (Bendell 2006). Replacement of natural shoreline vegetation with hard structures reduces filtration and nutrient transformation of runoff (Kemp et al. 2005), accumulation of fine sediments and organic matter (Snelgrove et al. 2000), habitat for shoreline-associated nekton (Jennings et al. 1999, Bilkovic and Roggero 2008), energy input to near-shore detritus based food webs at the land-water interface (Burkholder and Bornside 1957, Teal

1962), and dissipation of wave energy (Bendell 2006, Perry and Atkinson 2009). Intact shoreline vegetation normally supports abundant benthic communities also capable of nutrient transformation and trophic transfer through secondary production (Diaz and Schaffner 1990, Valiela 1995, Nixon and Buckley 2002). Despite the importance and ubiquitous nature of soft bottom habitats (Snelgrove 1999, Levin et al. 2001), shoreline development impacts on infauna remain under-studied.

An estimated 50 – 80% of the remaining wetland marsh habitat in Virginia may be lost due to sea level rise without shoreward progression of the marsh (Perry and Atkinson 2009), which may affect benthic species and their predators. Homeowners may harden their shorelines in response to sea level rise to prevent erosion, potentially stopping shoreward progression of tidal marshes and exacerbating their loss (Silliman and Bertness 2004, Perry and Atkinson 2009). The subsequent inability of tidal marshes to provide their valuable ecosystem services and perform their role in detritus production will likely impact infaunal benthos, especially key species such as the Baltic clam, *Macoma balthica*, a facultative deposit feeder (Lin and Hines 1994). Benthic infauna are widely recognized as the base of the food web in shallow systems within Chesapeake Bay (Baird and Ulanowicz 1989, Gillett and Schaffner 2009), and provide an important food source for the blue crab, with *M. balthica* composing up to 55% of adult blue crabs' diet (Hines et al. 1990, Seitz et al. 2001, Lipcius et al. 2007) (Figure 1). Therefore, shoreline development may be deleterious to one of the most valuable fisheries within the Bay (Kennedy et al. 2007).

Anthropogenic environmental disturbances, such as habitat destruction (Thrush et al. 2001), hypoxia (Dauer 1993), and toxic pollution (Gray 1979, Dauer 1993, Preston

and Shackelford 2002) reduce biological diversity and/or biomass at wide spatial scales in relatively low-structure, soft-sediment systems (Dauer 1993, Thrush et al. 2001, Preston and Shackelford 2002). In models of estuarine benthic succession after a disturbance, the number of individuals (density), biomass, and number of species (species richness) tend to change in different ways through ecological succession (Pearson and Rosenberg 1978, Rhoads et al. 1978, Newell et al. 1998). Biomass and diversity can be reduced in favor of opportunistic species that respond quickly after a disturbance, though the density of these organisms can increase rapidly and can be highly variable (Dauer 1993, Thrush et al. 2001). Opportunistic species in marine systems respond quickly by increasing their population densities after disturbances such as pollution (Pearson and Rosenberg 1978), hypoxia (Santos and Simon 1980, Dauer 1993), dredge spoil disposal (Rhoads et al. 1978, Newell et al. 1998), and sediment movement (Zajac and Whitlatch 1982, Commito et al. 1995). Known opportunistic infaunal species include spionid polychaetes (Rhoads et al. 1978, Santos and Simon 1980, Zajac and Whitlatch 1982, Quintino et al. 2006), clam worms (*Neanthes succinea*, Holland 1985), *Owenia fusiformis* (Rhoads et al. 1978), *Macoma mitchelli* (Holland et al. 1987), *Mulinia lateralis* (Rhoades et al. 1978, Santos and Simon 1980, Holland 1985), *Gemma gemma* (Commito et al. 1995), and gammarid and corophiid amphipods (Santos and Simon 1980, Newell et al. 1998). Capitellid polychaetes (e.g. *Capitella capitata*) are known to be opportunistic species in some systems (Pearson and Rosenberg 1978, Rhoads et al. 1978). However, in shallow mesohaline Chesapeake Bay, two non-opportunistic capitellid species are most abundant: *Mediomastus ambiseta* is the most abundant and exhibits relatively prolonged recruitment, low mortality (Diaz 1984, Holland et al. 1987), and sensitivity to pollution

(Weisberg et al. 1997), and *Heteromastus filiformis*, which is a long-lived eurytolerant species with a relatively large standing stock biomass (Holland et al. 1987).

Benthic communities exhibit high natural variability, and their density, biomass and diversity are correlated with salinity, sediment, latitude, physical energy, and depth (Boesch 1977, Dauer et al. 1984, Dauer et al. 1987, Holland et al. 1987, Schaffner et al. 1987, Diaz and Schaffner 1990, Snelgrove and Butman 1994). In the York River Estuary, the distribution and abundance of species is correlated with salinity, bottom sediment type, hydrodynamics, oxygen regime, and other variables (Schaffner et al. 2001, Gillett and Schaffner 2009). Benthic invertebrates are often restricted in mobility, which makes them ideal candidates for the study of changes in local environments, such as pollution (Gray 1979). As bottom type, hydrodynamics, and other correlates with benthic community structure are altered by shoreline development, infaunal distribution, density, and biomass are likely to be altered. Over the long term, shoreline development reduces biodiversity and abundance of benthic infauna (King et al. 2005, Seitz et al. 2006) and marsh-edge nekton and epifauna (Peterson et al. 2000, Jivoff 2005, Partyka and Peterson 2008). Sediment chlorophyll-a concentration, percent total organic carbon, sediment grain size and predator abundance varied by shoreline type (natural marsh, riprap, or bulkhead) in the Patuxent River, MD, and the benthic species assemblage differed by shoreline, likely driven by changes in the aforementioned variables (Bradley 2011).

Infaunal density, biomass, and diversity vary by different shoreline types, generally showing reductions adjacent to hardened shorelines. In a study of two sub-estuaries of the lower Chesapeake Bay, the Elizabeth-Lafayette River system and York River, there were reduced densities and diversities of subtidal infauna at shorelines

developed with bulkheads in both systems (Seitz et al. 2006). Specifically, abundance of *M. balthica* was significantly lower at bulkhead shorelines, intermediate at riprap, and higher at marsh shorelines in the Elizabeth-Lafayette, an urban, highly developed estuary, with over 50% of its shoreline developed (Seitz et al. 2006). In the York River, which is much less developed, abundance and Shannon diversity ( $H'$ ) of the entire benthic infaunal community was reduced adjacent to bulkhead shorelines compared to marsh or riprap shorelines (Seitz et al. 2006). In a study of the suitability of four types of shoreline as juvenile blue crab habitat, infaunal prey density was highest adjacent to marsh shorelines, intermediate adjacent to bulkhead, and lowest adjacent to riprap shorelines, where it was characterized by opportunistic species such as small polychaetes and nematodes (Long et al. 2011).

As shoreline development alters biodiversity and abundance of biota, predator-prey interactions and trophic transfer between these organisms are subject to stressors (Peterson et al. 2000, Carroll 2002, King et al. 2005). Blue crabs were at higher abundance in watersheds with high % marsh shorelines (King et al. 2005). Additionally, decreased abundance of blue crabs has been documented at riprap compared to natural marsh shorelines (Carroll 2002). A study of both infauna and epifauna in the Pascagoula River estuary, MS, examined habitat conditions across a gradient of shoreline development that considered two factors: level of total alteration as low, medium, and high zones, and adjacent shoreline condition as “restricted” (*i.e.* hard structures, spoil beaches or leveed banks) or “unrestricted” natural shorelines (Partyka and Peterson 2008). Infaunal density and taxa richness do not differ significantly by total alteration zone or adjacent shoreline condition, though sites with adjacent unrestricted shorelines tend to

have higher abundance than those with adjacent restricted shorelines. Epifaunal density and richness, however, are significantly greater at sites with adjacent unrestricted than restricted shorelines, and density is highest in the low-development zone, intermediate in the medium-development zone, and lowest in the high-development zone (Partyka and Peterson 2008). In a cluster analysis, there is close association between infaunal communities in medium-development zones adjacent to restricted shorelines and communities adjacent to both types of shorelines in high-development zones. On the contrary, there are close associations in epifaunal communities between adjacent unrestricted shorelines from medium- and low-development zones (Partyka and Peterson 2008). Thus, responses to shoreline development may depend on the amount of undeveloped shoreline in the neighboring area. In the James River, VA, fish communities respond to shoreline condition and upland land use (Bilkovic and Roggero 2008). Fish community index (a multimetric index comprised of species richness/diversity, abundance, trophic composition and species composition) is reduced adjacent to bulkhead in comparison to marshes and riprap shorelines (Bilkovic and Roggero 2008).

Given the prevalence of hardened shorelines within Chesapeake Bay and the evidence linking altered shoreline habitats to reduced infaunal and predator community metrics, it is essential to continue to examine the effects of shoreline development on the infaunal benthos and its predators. Previous work has examined patterns for existing shorelines, but few studies have examined changes before and after development.

### ***Living Shorelines: Alternatives to Conventional Shoreline Protection***

Shoreline development does not always have to result in hardening and destruction of the land-water interface and intertidal zone. Shoreline stabilization

methods that incorporate as many natural habitat elements as possible, while still protecting shorelines from erosion are called “living shorelines”.

Living shorelines represent an alternative to conventional shoreline armoring methods (bulkheads, seawalls and riprap revetments) for reducing and preventing erosion, and are highly variable and tailored to a specific location. Natural habitat elements used in living shorelines include marsh grasses, submerged aquatic vegetation, riparian vegetation, coarse woody debris, and oyster reef and shell, particularly as formed into offshore breakwaters to mimic natural reefs that have largely disappeared from Chesapeake Bay (Erdle et al. 2006). Use of natural elements for shoreline erosion control are hypothesized to perform as well as, if not outperform, conventional methods, in addition to providing better water quality habitat functions for near-shore organisms (Erdle et al. 2006), especially when used in lower-energy environments (Smith 2006).

Beach nourishment, dune restoration, tidal marsh creation and enhancement, bank grading and fiber logs are also nonstructural elements used in living shorelines (Duhring 2006). Hybrid living shorelines include rock and natural elements in combination (*e.g.* marsh edge stabilization or marsh toe revetments, marsh sills, marshes with groins, and offshore breakwaters), and have particular effectiveness in higher energy environments (Duhring 2006, Smith 2006). Rock is often placed offshore, as part of a breakwater or sill oriented to wind and waves, to dampen wave energies and maintain or allow for the development of marshes and beaches behind them (Smith 2006). Breaks in sills are often intentionally added to allow for tidal flushing and connectivity of the shoreward water to the channel-ward water; these breaks are called gaps or tidal gates. While living shorelines may not provide as much erosion protection as conventional hard structures,



they allow for mobility of the shoreline and near-shore sediments and act as part of the natural system rather than against it. In so doing, these shorelines are likely to improve in effectiveness over time as planted marsh grasses grow and stabilize sediment, oyster shell breakwaters become covered with live oysters, and sedimentary processes stabilize (Smith 2006).

Living shorelines are dynamic structures, thus, building effective shorelines capable of erosion control requires a good deal of skill and planning, more so than conventional strategies, and even a single failure can be detrimental to public acceptance (Smith 2006). In Virginia, examination of the effectiveness of these procedures has been limited since they are still in their infancy. Nonetheless, structures placed along the edges of natural fringing marshes adjacent to low banks and marshes planted in spring were the most effective methods of structural and nonstructural (respectively) erosion control at study sites built within the past 10 years (Subramanian et al. 2006). In Maryland, study of living shorelines has been much more extensive, and 32 of 35 projects in Talbot County, including marsh sills, groins and marsh edge stabilization, showed improvement in erosion control and habitat; 83% of banks inspected were stable and 74% of marshes exhibited minimal or no erosion. For all projects, diligence by landowners and contractors for inspections and repairs was necessary to ensure function (Subramanian et al. 2006). In Maryland, a 2008 law now states that “living shorelines” are the preferred method for controlling erosion as they “trap sediment, filter pollution, and provide important aquatic and terrestrial habitat” (MD House of Delegates 2008).

In a before-after control-impact (BACI) study of a bulkhead site modified with installation of a living shoreline, densities of some marsh-associated organisms

(mummichog: *Fundulus heteroclitus*, grass shrimp: *Palaemonetes pugio*, and pumpkinseed: *Lepomis gibbosus*) increased after just two months, indicating that some organisms can respond immediately to restoration with living shorelines and suggesting that multiple structural habitat elements may help expand the ecological function of living shorelines (Davis et al. 2006). Additionally, four structural habitat types (riprap, oyster shell, vegetation and woody debris) host different suites of species, with blue crabs best suited to oyster reefs, and older life-history stages of organisms better suited to riprap habitats (Davis et al. 2006). As study of the ecological roles of living shorelines is currently limited, continued study will help to identify which aspects of living shoreline construction are best for both erosion control and support of near-shore communities (National Research Council 2007).

While placement of living shorelines does constitute a type of anthropogenic development of the shoreline, their use is intended to provide the best habitat condition possible for the near-shore community while still maintaining erosion control. Given this intention of living shorelines to obtain to a more natural state, and the inclusion of one living shoreline change in this project, shoreline development will hereafter be referred to as "shoreline modification" so as to remove the connotation that development must always result in shoreline hardening.

### ***Quantitative Methods***

To reliably determine the degree to which shoreline modification impacts near-shore communities, a BACI, or before-after control-impact design was employed in this study. This method is widely used in determining environmental impacts on populations caused by anthropogenic disturbances (Underwood 1992). BACI sampling involves

collection of samples before a disturbance at both the future impacted site and control sites that are assumed to remain constant with respect to the impact. Shoreline modification projects are particularly well suited to be studied with this framework because they are planned disturbances. In Virginia, permits are required before beginning a project; full plans are made and reported to the Wetlands Board, the Virginia Marine Resources Commission (VMRC), and the Department of Environmental Quality (DEQ), and proposed shoreline modifications undergo public interest review, site visits, and hearings before permission is granted to build (Watkinson and Moon 2006).

In the BACI sampling framework, an anthropogenic disturbance in the impacted location (*i.e.* modified shoreline) is likely to cause a different pattern of change from that of the temporal variation in the control (*i.e.* unchanged shoreline) location. The difference is detected as a significant interaction term in the statistical analysis; in an ANOVA, this would be an interaction between the main effects of time (*i.e.* before and after) and treatment (*i.e.* control and impacted shorelines; Underwood 1992). In a linear modeling framework with model comparison, this would be an interaction that is not only included in a well-supported model, but has a significant parameter estimate (Johnson and Omland 2004), and whose sign indicates the direction of response to shoreline modification.

To improve the ability to detect inter-annual variation from temporal change in a potentially impacted location, an asymmetrical design with a randomly selected set of control locations can be used. In this study, two control sites (with replicates) were included for each study location. Thus, a significant impact must cause a temporal change that is different from what is expected based on the multiple control locations,

preventing the interpretation of inter-annual variation as an impact of shoreline modification.

Samples in an ideal BACI framework are taken at replicated, random intervals of time before and after the impact starts to ensure that chance temporal fluctuations in either location do not confound the detection of an impact. Changes in the environment can be perceived as disturbances, and can be categorized in two ways: pulse and press disturbances. Pulse disturbances result from short-duration impacts, such as disposal of dredge material (Schaffner 2010) or an accidental spill of a toxin (Pearson and Rosenberg 1978), after which population, community, or ecosystem metrics return to their original values (Underwood 1992). Press disturbances result from sustained impacts, such as the placement of a dam, persistence of heavy metal pollutants, or climate change (Lake 2000, Schiel et al. 2004) after which population, community, or ecosystem metrics do not return to their original values but remain constant at a higher or lower value (Underwood 1992, Lake 2000). Differences can be detected for pulse and press impacts through the expression of different patterns of significant interactions between times and locations of sampling (Underwood 1992, 1994, Underwood et al. 2003). If a significant interaction only occurs at a single time frame, a pulse disturbance is likely; in contrast, if significant interactions continue to occur in time, a press disturbance is likely.

Current work has tended to focus on metrics of communities that do not examine particular species within the community using a BACI study, and thus, might be missing elements of ecological succession as communities respond to shoreline modification stressors. Most studies do not examine species-specific responses, with the exception of studies that include multivariate analyses. Multivariate analyses tend to be dominated by

cluster analysis (King et al. 2005), ANOSIM with nMDS or ordination (Chapman and Bulleri 2003, Seitz et al 2006, Bilkovic and Roggero 2008, Partyka and Peterson 2008), which effectively cluster communities that respond similarly to shoreline changes and are able to detect species-specific differences. I was only able to find two studies that examined species-specific responses of infauna to shoreline changes over time. In one study, there was an increase in filter feeders after shoreline modification with bulkhead, but this study did not include control sites without modification (Ahn and Choi 1998). In the other study, there were higher abundances of oligochaetes and nematodes at natural shorelines through time, but this was not a before-after study (Sobocinski et al. 2010). In light of this, I have examined species-specific community structure, examining differences among species within the BACI sampling design.

### ***Study Objectives and Hypotheses***

In this study, I investigated the impacts of shoreline modification on near-shore communities using a temporal and spatial approach. The specific objectives of the study were three-fold. 1) I examined the direct effects of three different types of shoreline modification on benthic infaunal density, biomass, and diversity immediately before and one year after completion of the projects, using a before-after control-impact, or BACI study design. 2) I examined infaunal community composition by groups of the most common taxa and by the lowest possible taxonomic classification (typically species) to document changes in community composition with time and shoreline treatment. 3) I examined effects of shoreline modification on near-shore predator abundance (blue crabs and shoreline-associated fishes) and diversity (shoreline-associated fishes) at each of the BACI study sites.

I hypothesized that infaunal abundance, biomass, and diversity would differ from 2010 to 2011 sampling periods between control and impact sites as a result of changes due to shoreline modification. Based on the empirical examples outlined in the introduction, I expected the three community response variables (infaunal density, biomass, and diversity) to decline in three study locations changing from undeveloped to developed shorelines (Holly Cove, VA; Timberneck Creek, VA; and Dandy, VA). In contrast, I expected the three community response variables to increase at one study location changing from bulkhead to living shoreline (Windy Hill, MD). Based on previous literature, I expected these changes to be predicted most by shoreline treatment and sediment grain size. Additionally, I expected predator density to vary similarly to benthic infauna, as soft-sediment systems in Chesapeake Bay tend to be structured by bottom-up control (Seitz and Lipcius 2001).

## **METHODS**

### ***Study Locations***

Using permitting data for the construction and removal of structures that altered shoreline and near-shore habitats, four locations in the Chesapeake Bay (Figure 2) were chosen based on their timing of construction, accessibility, and length of impacted shoreline. The study locations were restricted to areas developed in summer 2010 within Chesapeake Bay that were accessible by boat or trailer in one day trip from Gloucester Point, VA, were no more than 1.2 m deep 5 m from the shoreline at mid-tide, and included at least 30 m (preferably longer) of impacted shoreline length.

Three sites were changed from undeveloped to developed (riprap, boat slips) shorelines, and one site was changed from developed (bulkhead) to living shoreline (Figure 3). Timberneck Creek, VA, contained a natural marsh and tidal wetland shoreline that was developed with boat docks and slips (84 m in linear length along the shoreline) and three discontinuous sections of riprap (sections are 81, 37, and 26 m in length). Dandy, VA, contained a *Phragmites australis* (invasive common reed) marsh shoreline that was replaced by riprap of 49 m linear length. Holly Cove, VA, contained a tidal wetland shoreline that was replaced by riprap of 34 m linear length. Windy Hill, MD, contained a section of bulkhead (149 m in length) that was removed and replaced by a living shoreline consisting of an offshore breakwater, marsh sills, and planted marsh grasses. The length of the living shoreline is now 309 m when measured around the breakwaters and along the perimeters of the coves between the breakwaters.

### ***BACI Sampling***

To understand the impacts of shoreline modifications, it is important to directly

measure the faunal changes that occur over the time frame in which modifications occur. A before-after control-impact (BACI) study design (Stewart-Oaten et al. 1986, Underwood 1992, 1994, Underwood et al. 2003) was used to examine changes in community structure due to shoreline modification at each of the four study locations. The BACI study design involves sampling at impact sites (subjected to shoreline change) sites and similar, paired control sites, both before and after the impact (Underwood 1992, 1991, Hewitt et al. 2001). Samples at nearby control groups should respond similarly to temporal fluctuations, whereas impact shorelines may start out similar or different to control sites, but respond differently in 2011, after shoreline modification than control shorelines (Figure 4). Replicate samples were taken before construction in May-June 2010 and again in May-June 2011, after construction was complete (Table 1). This is an important advance in shoreline modification studies, as it provides a method to account for many site-specific differences (such as wave influence, intra-annual variation, and differences in physical parameters) prominent in data sets of comparative studies by comparing fauna at individual sites before and after a change (Stewart-Oaten et al. 1986, Underwood 1992, Underwood et al. 2003).

BACI sampling schemes can also benefit from asymmetrical designs, that is, more than one control group, and one or more impact groups (Underwood 1992). More often than not, environmental impacts are not replicated in nature, but that does not prevent the inclusion of multiple controls with which to compare an environmental impact. This strategy of assigning multiple controls allows the researcher to increase the likelihood of detecting an impact in a variable environment subjected to disturbances in addition to temporal heterogeneity (Underwood 1992). This way, multiple control



groups may differ from one another, capturing spatial variation, but should respond similarly to temporal change, whereas impact shorelines may respond differently after shoreline modification. Thus, having multiple control groups is another way to account for variability within the control observations, which is beneficial given the high degree of variability and patchiness in benthic communities with salinity, sediment, hydrodynamics, latitude and depth (Boesch 1977, Dauer et al. 1984, Dauer et al. 1987, Holland et al. 1987, Schaffner et al. 1987, Diaz and Schaffner 1990, Snelgrove and Butman 1994). In this experiment, two control sets were grouped separately at each study site, blocked by close geographical proximity or shoreline type. Sites changing from undeveloped to developed shorelines (Timberneck, Dandy, and Holly Cove) had controls grouped by geographical proximity, as the impact shoreline started with the same shoreline type as other shorelines in the area. The site changing from bulkhead to living shoreline (Windy Hill) had controls blocked by shoreline type, as the shoreline started as bulkhead and changed to a more natural type shoreline, so the controls were assigned to resemble both the before (bulkhead) and after (marsh) conditions of that impacted shoreline. Impact shorelines were sampled as two separate blocks at Timberneck Creek, given the differing nature of the impacts (one changed from tidal wetland and marsh to boat docks), whereas impact shorelines at all other study sites were sampled as one block.

Numbers of replicate samples at each site are slightly different due to constraints of time and space (Table 1). For example, infauna and crab scrape samples were collected in one day at Windy Hill, MD and Holly Cove, VA, to reduce travel costs, thus limiting the number of samples that could be collected. The length of impacted shoreline

available for sampling differed at each site, leading to more samples taken adjacent to longer shoreline change areas (Table 1). The field time required to process seine samples put a constraint on seine sampling, leading to fewer seines than other sample types at all sites (Table 1).

### *Infauna*

To examine benthic infauna, two types of samples were taken at each site, one for larger, deep-dwelling macrofauna, and another for shallower macrofauna. At least two and up to 12 replicate infaunal sample pairs were taken at each shoreline treatment in each year (Table 1). To collect deep-dwelling infauna, a large PVC core 38 cm in inner diameter was pushed 40 cm into the sediment and evacuated with a suction apparatus (as in Seitz et al. 2006) into a 3-mm mesh bag, collecting a sample 0.113 m<sup>2</sup> in surface area (Eggleston et al. 1992). Samples were bagged and placed on ice immediately and transported back to the lab where they were frozen until processing. In the lab, these samples were sorted, transferred to 70% ethanol, enumerated, identified to the lowest possible taxonomic level, measured with calipers to the nearest 0.1 mm and biomassed to ash-free dry weight in a muffle furnace at 550°C for 4 hours. Preservation of invertebrates in ethanol does not have any different effects on biomass estimates than preservation in formalin (Wetzel et al. 2005).

To collect smaller macrofauna, a hand-held PVC core 10 cm in inner diameter was pushed 15 cm into the sediment and sieved in a 500-µm mesh bag, collecting a sample 0.008 m<sup>2</sup> in surface area. All animals from these 500-µm samples were fixed in normalin for at least 24 hours, then sieved again on a 500-µm mesh screen, transferred to

70% ethanol, sorted, enumerated, identified to the lowest possible taxonomic level, and biomassed to ash-free dry weight by incineration in a muffle furnace at 550°C for 4 hours.

At each infaunal sampling site, surface salinity, dissolved oxygen (DO), and water temperature were measured using a calibrated YSI Pro-Plus Multi-Parameter Water Quality Meter. Additionally, two sediment samples were taken at each site with a 2.6-cm diameter syringe core pushed into the sediment to a depth of 5 cm. These samples were used to analyze grain size using a standard wet sieving and pipetting technique (Plumb 1981).

Due to logistical constraints, a subsample of replicates for the 500- $\mu$ m samples was sorted. For Timberneck, a random number generator was used to select the first 11 (2010) or 10 (2011) samples from all control groups, and the first four samples from either impact groups for both years. Subsampling provided insufficient sample sizes for analysis of each impact separately, preventing them from analysis as separated impacts, so they were combined. Control groups were each well represented and were left separated. For Dandy, Holly Cove, and Windy Hill, a random number generator was used to subsample 500- $\mu$ m infauna by selecting two impact shoreline samples from each year, and two control shoreline samples from one of the control groups for each year. The control group from which samples were pulled was decided by whichever sample was chosen first, *i.e.* if a sample from control group 2 was selected first, only samples from control group 2 were sorted and included in the study.

### *Crab Scrapes*

In addition to infaunal samples, epifaunal predator samples were collected at each location, targeting blue crab predators. At least two and up to 12 scrapes were taken at

each shoreline treatment in each year (Table 1). The sampling device used was a “crab scrape”, which is a crab dredge 1 m in width modified with teeth removed and skis placed on the lowered side of the dredge. The scrape was lowered from a small boat, and dragged along the bottom along the shoreline for 20 m (measured using a GPS), leading to samples 20 m<sup>2</sup> in area. A bag with approximately 7-mm mesh was attached to the scrape to collect crabs with  $\geq 10$ -mm carapace width as well as fish. All crabs and fish were collected from each scrape, identified to the lowest taxonomic level (typically species), enumerated, measured (standard length) to the nearest mm, and released on site.

### *Seines*

Near-shore fishes and blue crabs were collected using beach seines 15.24 m in length. At least two and up to six seines were taken at each shoreline treatment in each year, except in 2010 at Dandy, where the construction began early and before our predator samples could be taken (Table 1). Seines were deployed according to the methods of the VIMS Seine Survey ([www.fisheries.vims.edu/trawlseine/sbintro.htm](http://www.fisheries.vims.edu/trawlseine/sbintro.htm)), in which one person anchors one end of the net at the shoreline. Another person then walks the other end straight out perpendicular to the shoreline as far as possible, and sweeps in a quarter-circle arc motion back towards the shoreline, resulting in a sample 182.4 m<sup>2</sup> in area. All fishes and blue crabs were collected from each tow, identified to the lowest taxonomic level possible (typically species), enumerated, total length measured to the nearest mm, and released on site, as per approved IACUC regulations; permit # IACUC-2010-05-10-6718-rdseit). Individuals that could not be positively identified were placed in ice slurry and taken back to the lab for species confirmation, and about 10 specimens of each species were kept for future gut-content analysis from each seine. Dr. Denise

Breitburg's Lab at the Smithsonian Environmental Research Center (SERC) in Edgewater, MD, collected seine samples at Windy Hill in 2010 using both quarter-arc and parallel seining methods. To compare these seines with those taken by our lab group, abundances were normalized by total area seined.

### ***Data Analysis***

#### *Assessment of Linear Modeling Assumptions*

Before model construction and analysis, all response variables were examined to assure their accordance with assumptions of linear models: normality, homogeneity of residual variance, and independence of predictor variables. Normality was assessed through visual interpretation of QQ plots and histograms, and homogeneity of residual variance (homoscedasticity) was assessed through visual interpretation of Residuals vs. Fitted and Scale-Location plots as well as a Brown-Forsythe Test (Brown and Forsythe 1974, Logan 2010). If assumptions for normality or homoscedasticity were not met for abundance or biomass data, the response variable was transformed using Box-Cox transformation, and analyses run on transformed data. All diversity data were transformed using the logit transformation, as the raw data are constrained between 0 and  $\log(S)$ , and are thus not normal (Warton and Hui 2011).

To examine the possibility of collinearity of the environmental predictor variables, DO (mg/L), Salinity (PSU), and Temperature ( $^{\circ}\text{C}$ ), I examined the correlation structure of those three variables over all four sites, visually assessed a scatter plot matrix (“splom”) of the three variables regressed against one another, and calculated tolerance and variance inflation factors, hereafter VIFs (Logan 2010, R.J. Latour pers. comm.). Tolerance values below 0.2 and VIF values above 5 are considered to be evidence of

collinearity (Logan 2010, R.J. Latour pers. comm). For salinity, the tolerance value was the lowest (0.49) and the VIF value the highest (2.02) of all three predictor variables. While these values are within an acceptable range, I decided they were high enough to support my choice of using salinity as a single predictor variable. Salinity is a major determinant of infaunal distribution patterns in estuaries (Boesch 1977, Barnes 1989, Diaz and Schaffner 1990, Llansó 2002). In addition, sampling during a given day was not randomized among control vs. impact treatments (*e.g.*, samples were taken adjacent to all impact sites, then all controls, based on proximity to the next sample), and DO and temperature tend to vary more by time of day than does salinity. Thus, salinity is the most robust choice for a single environmental predictor value.

Independence of predictor variables for individual sites was also assessed through visual examination of sploms and by calculating VIFs (Logan 2010, R.J. Latour pers. comm.). Models with collinear variables were removed from the model sets before model comparison.

#### *Linear Modeling and AIC*

Linear models were used to model infaunal response variables (abundance, biomass, and diversity) for the 3-mm size-class data at all study sites, as well as the 500- $\mu\text{m}$  size class at Timberneck Creek only. Models were constructed with a combination of several explanatory variables, or parameters, in addition to the year and shoreline treatment effects of the BACI design (Table 2).

The explanatory parameters included in different combinations in the linear models are year (“Year” or “Y”, 2010 or 2011), shoreline treatment (“Treat” or “T”, control 1, control 2, or impact), an interaction term of year and shoreline treatment ( $Y \times T$ ),

sediment grain size as percent of sand + gravel (“Sed”), and salinity in PSU (“Sal”). Impact samples in the 3-mm size class were assigned to either impact 1 or impact 2, depending on whether they were taken at boat slips and docks (impact 1) or riprap (impact 2) shorelines. Impact samples in the 500- $\mu$ m size class were all assigned to one impact group, since subsampling provided insufficient sample sizes for analysis of each impact separately.

Linear modeling is the basis for the ANOVA analysis framework, for which a BACI study assigns year and treatments, in this case different shorelines. In this study, multiple linear models were employed to analyze the 3-mm size class because additional parameters were included in the candidate models to account for some of the variance in the response variables not attributed to year and shoreline treatment. The efficacy (probability of best describing the data) of multiple candidate models that contain different combinations of parameters can be compared using model comparison (*e.g.* AIC). In this study, year (“Y”) and shoreline treatment (“Treat”) are categorical, or “dummy” predictor variables, analogous to the “main effects” in an ANOVA comparison of means. In addition, sediment grain size (Sed) as % coarse sediments (a combination of gravel and sand fractions), and salinity (Sal) were selected as explanatory variables.

For each response variable (abundance, biomass, and diversity), a set of proposed candidate linear models was developed (Tables 2, 3), and the relative efficacy of these models for each response variable was analyzed using an information-theoretic approach with Akaike’s Information Criterion, or AIC (Johnson and Omland 2004, Anderson 2008). This approach allows the user to build models to represent the information “loss” when a model approximates reality (Gotelli and Ellison 2004, Johnson and Omland 2004,

Anderson 2008) and allows for model comparison and selection, particularly useful in observational studies where uncontrolled variables exist (Johnson and Omland 2004, Stephens et al. 2005, Anderson 2008).

The relative strength of each model of the set is determined by its AIC value corrected for low sample size (Equation 1). Within the equation below,  $n$  represents the sample size,  $k$  the number of parameters included in each candidate model, and  $\sigma^2$  the residual sum of squares divided by  $n$  (from least squares regression) which can be used as a proxy for log-likelihood, and is typically used in AIC calculations.

$$\text{Equation 1.} \quad AICc = n * \ln(\hat{\sigma}^2) + 2k \left[ \frac{n}{n - k - 1} \right]$$

Next, delta  $AIC_d$  values are calculated to compute simple differences between the best-fitting model and the rest of the models (equation 2), representing the expected information loss between the best model and the  $i^{\text{th}}$  model as compared to the model with the lowest AICc value ( $AICc_{\min}$ ).

$$\text{Equation 2.} \quad \Delta_i = AICc_i - AICc_{\min}$$

From these delta values, a discrete weighted probability, or Akaike weight, is calculated to determine the model with the highest probability of being that which best fits the data (equation 3). In the equation below, the numerator represents the model of interest's likelihood, and the denominator represents the sum of all other model likelihoods. An Akaike weight ( $w_i$ )  $\geq 0.10$  indicates strong support of a model (Anderson 2008), as it best describes the dataset. Hereafter, these models will be referred to as "best-supported" models.

$$\text{Equation 3.} \quad w_i = \frac{\exp(-\frac{1}{2} \Delta_i)}{\sum_{r=1}^R \exp(-\frac{1}{2} \Delta_r)}$$



To examine the absolute support for the model (not in comparison to the other models, but in comparison to the data itself), adjusted  $R^2$  values, which provide a penalty for unnecessary parameters, were calculated for all models in each candidate set. Individual parameter estimates for the best-supported models were examined to determine important predictor variables for each response.

Within the BACI framework, if a model that included an interaction term between year and shoreline treatment ( $Y \times T$ ) had a high Akaike weight, and the estimated beta coefficient ( $\beta$ ) of that interaction term was significant (that is, the confidence interval associated with its standard error did not overlap with 0), there is evidence to say that the year and shoreline treatment interaction was significant.

A significant interaction between year and shoreline treatment indicates an effect of shoreline treatment, as long as it compared control to impact shorelines (Underwood 1992, Underwood 2003), and was designated by 80% confidence intervals round the mean that do not overlap 0 (Underwood 1997). This significant interaction term would show that the response variable (*e.g.* abundance) was affected differently at control versus impact shoreline treatments in 2011, after the shoreline modification project has been completed. For example, if abundance were to decrease at the impact shoreline treatment but increase or remain constant at both control shorelines, and the interaction term between year and treatment was significant, this would indicate that the change in abundance was due to the change in shoreline condition (Underwood 1992, 1994, Underwood et al. 2003). However, if the interaction term between the two control shoreline treatments were significant, this would indicate a highly variable data set and an effect of shoreline modification could not be inferred.

This method is more desirable than analyzing response variables with a single model, as it allows the user to develop a set of competing, biologically relevant hypotheses of different combinations of parameters (Johnson and Omland 2004), provide evidence for each model, and examine those parameters that are most important in modeling the response variable (Johnson and Omland 2004, Anderson 2008). Therefore, even if a statistically significant interaction term between year and shoreline treatment is found in a model, if there is no evidence that the model fits the data, the interaction is not important to the response variable.

#### *Subsampled Datasets*

At Dandy, Holly Cove, and Windy Hill, the subsampling procedures on the 500- $\mu\text{m}$  data set resulted in two replicates, preventing the ability to analyze these data reliably using linear models that include additional predictor variables. Instead, these data were analyzed with two-way ANOVA, using year and shoreline treatment as the factors. Since ANOVA is a special case of linear models with only categorical predictor variables, this is analogous to the model comparison of 3-mm data set, but without the inclusion of additional predictor variables and model comparison that require additional degrees of freedom.

Before ANOVA analysis, all response variables were examined to assure their accordance with the assumptions of ANOVA. Normality was assessed through visual interpretation of QQ plots and histograms, and homogeneity of variance (homoscedasticity) was assessed through visual interpretation of Residuals vs. Fitted and Scale-Location plots as well as a Brown-Forsythe Test (Brown and Forsythe 1974, Logan 2010). If assumptions were not met for abundance or biomass data, the response variable

was transformed using Box-Cox transformation, since it consistently performed best as a transformation for the response variables of the 3-mm data set, and allows the transformation to be derived from the data itself (R.J. Latour, pers. comm.). All diversity data were transformed using the logit transformation, as they represent data constrained between 0 and  $\log(S)$ , and are thus not normal (Warton and Hui 2011).

Analyses were completed and figures developed using R statistical and graphing software (version R-2.15.1.tar.gz; <http://cran.r-project.org>), and several packages (car, ggplot2, HH, MASS, MuMIn, plyr, stats) on the R-studio platform (0.96.331; <http://rstudio.org>). Individual analyses were conducted for each response variable: infaunal density, diversity, and biomass.

## RESULTS

### *Water Quality and Sediment Grain Size*

Salinity, DO, and temperature did not vary greatly between treatments, but differed between years at the four sites (Table 4). At Timberneck, salinity was slightly higher in 2010 than in 2011. Neither of the other two variables differed considerably among shoreline treatments or from 2010 to 2011 (Table 4). At Dandy, salinity and DO decreased while temperature increased at all shoreline treatments from 2010 to 2011, though there were no notable differences by shoreline treatment (Table 4). At Holly Cove, salinity, DO, and temperature decreased at all shoreline treatments from 2010 to 2011, though there were no notable differences by shoreline treatment (Table 4). At Windy Hill, salinity decreased slightly, DO decreased markedly (by over half in some areas), and temperature increased from 2010 to 2011, though there were no notable differences by shoreline treatment (Table 4).

Substrate grain size encompassed fines (silt and clay,  $< 62.5 \mu\text{m}$  diameter), sands ( $62.5 \mu\text{m} \leq \text{diameter} \leq 2,000 \mu\text{m}$ ), and gravel ( $>2,000 \mu\text{m}$  diameter). Across all treatments at Timberneck, notably, mean % sand increased from 55.70 in 2010 to 66.83 in 2011. From 2010 to 2011, mean % clay decreased from 15.48% to 10.92%, mean % silt decreased from 28.55 to 20.94, and mean % gravel increased from 0.27 to 1.31. Trends in mean sediment grain size were driven by increases in fine grains (clay and silt) and decreases in coarse grains (sand and gravel) at impact 2, with both control and impact 1 shorelines remaining relatively constant (Figure 5). Across all treatments at Dandy, notably, mean % sand decreased from 94.86 in 2010 to 89.51 in 2011 and mean % gravel increased slightly from 0.85 in 2010 to 3.17 in 2011. From 2010 to 2011, mean % clay

remained virtually constant (1.94 to 2.02), and mean % silt remained virtually constant (2.35 to 2.33). The main trends were driven by: (1) an increase in clay at control 1 and impact, but decrease at control 2 shorelines, (2) an increase in silt at control 1, but decrease at control 2 with little change at impact shorelines, (3) a decrease in sand at all treatments from 2010 to 2011, and (4) an increase in gravel at all shoreline treatments (Figure 5). Across all treatments at Holly Cove, mean % gravel increased from 8.32 in 2010 to 13.21 in 2011. From 2010 to 2011, mean % clay remained relatively constant (2.14 to 1.78), mean % silt decreased slightly (10.31 to 8.89), and mean % sand decreased slightly (79.23 to 76.13). The main trends were driven by and reflect changes at impact sites, with little change at control shorelines (Figure 5). At Windy Hill, mean % silt increased substantially from 1.47 in 2010 to 8.34 in 2011, and mean % sand decreased from 65.98 in 2010 to 58.09 in 2011. From 2010 to 2011, mean % clay tripled from 0.99% to 2.99%, and mean % gravel remained nearly constant (31.57 to 30.58). These trends were driven by: 1) increases in clay and silt at control 2 and impact shorelines, 2) decreases in sand at control 2 shorelines, 3) decreases in gravel at impact shorelines, and 4) little change at impact 1 shorelines (Figure 5).

### ***Infauna***

Benthic infauna were characterized by patchy distributions, including several samples with no infauna, and a few dominant taxa that varied depending on the site.

### ***Timberneck***

Density of 3-mm infauna at Timberneck Creek increased from 2010 to 2011 at both impact 1 (boat docks) and impact 2 (riprap), while staying constant at both control (Timberneck Creek and Catlett Island marshes) shorelines (Figure 6A). However, the

single best-supported model for Box-Cox transformed density was that including year and treatment without interactions ( $g_2$ : Y+T;  $w_i = 0.875$ ; Adj.  $R^2 = 0.1480$ ; Table 5). Density at control 2 and impact 1 shorelines differed significantly from the reference (control 1) based on their parameter estimates (Table 6), though interactions between year and treatment were not estimated by model  $g_2$ .

Biomass of 3-mm infauna at Timberneck increased at impact 1 (boat docks), while remaining virtually constant at both control shorelines and at impact 2 (riprap) shorelines (Figure 6B). The best-supported models for Box-Cox transformed 3-mm biomass were the global model ( $g_1$ : global;  $w_i = 0.619$ ; Adj.  $R^2 = 0.4676$ ) and that including year and treatment with interactions ( $g_4$ : Y×T + Sed;  $w_i = 0.327$ ; Adj.  $R^2 = 0.4407$ ), both of which had higher Adj.  $R^2$  values than models for density (Table 5). For biomass modeled by  $g_1$ , sediment grain size was an important predictor and impact 2 shorelines differed significantly from the reference (control 1) based on parameter estimates (Table 6), though interactions between year and treatment were not significant. Biomass modeled by  $g_4$  did not differ significantly between shoreline treatments, but sediment grain size was an important predictor, and the interaction between year and impact 1 (compared to the reference, control 1) was significant and positive based on parameter estimates (Table 6), indicating a positive effect of shoreline modification on biomass.

Diversity of 3-mm infauna at Timberneck, in contrast to biomass, increased at both control shorelines and impact 2, but stayed consistent at impact 1 (Figure 6C). The best-supported models for logit-transformed 3-mm diversity were that for sediment alone ( $g_6$ : Sed;  $w_i = 0.485$ ; Adj.  $R^2 = 0.2931$ ), that including year and treatment with

interactions and sediment grain size ( $g_4$ :  $Y \times T + \text{Sed}$ ;  $w_i = 0.409$ ; Adj.  $R^2 = 0.3971$ ), and the global model ( $g_1$ : global;  $w_i = 0.105$ ; Adj.  $R^2 = 0.6159$ ; Table 5). For diversity modeled by  $g_6$ , sediment grain size was an important predictor (Table 6). Sediment grain size was also an important predictor for diversity modeled by  $g_4$  and  $g_1$ , and the interaction between year and impact 1 was significant and negative based on parameter estimates (Table 6), indicating a negative effect of shoreline modification on diversity. Across response variables at Timberneck for 3-mm infauna, biomass and diversity were reasonably well explained by the candidate models, with high Adj.  $R^2$  values (between ~0.4 to 0.6). Abundance was not very well explained by the candidate model set, with relatively low Adj.  $R^2$  values ( $< 0.2$ ; Table 5).

Regarding community composition of 3-mm infauna by common taxa, bivalves decreased at both controls and impact 2, but increased at impact 1 from 2010 to 2011 (Figure 6D). Polychaete density increased at all shorelines, but at impact 2 it increased by a factor of almost 17 compared to increases by factors of 1.5 to 3.5 at the other shorelines (Figure 6D). Regarding species-specific community composition of 3-mm infauna, *Macoma balthica* drove the decrease in bivalve density at both controls, and *M. balthica* and *Macoma mitchelli* drove the decrease of bivalve density at impact 2 (Figure 6E). Additionally, *M. balthica* also drove the increase in bivalves at impact 1. *Aligena elevata* were lost at impact 1, but replaced by *M. balthica* and *Tagelus plebeius*. *Clymenella torquata*, *Neanthes succinea*, and capitellids drove the increase of polychaetes at impact 1, and capitellids drove the increase at impact 2 (Figure 6E). *Drilonereis longa* recruited to all shorelines, but their densities were lower at both impact shorelines.

Density of 500- $\mu\text{m}$  infauna at Timberneck Creek decreased from 2010 to 2011 at impact shorelines and control 1 shorelines (within Timberneck, the system where the impact occurred), but remained constant at control 2 (in the Catlett Islands, outside of the impact area) (Figure 7A). The best-supported models for density (untransformed) were that including year and treatment with interactions ( $g_3$ :  $Y \times T$ ;  $w_i = 0.381$ ; Adj.  $R^2 = 0.5728$ ), year and treatment with interactions and sediment grain size ( $g_4$ :  $Y \times T + \text{Sed}$ ;  $w_i = 0.302$ ; Adj.  $R^2 = 0.6029$ ), and year and treatment with interactions and salinity ( $g_5$ :  $Y \times T + \text{Sal}$ ;  $w_i = 0.227$ ; Adj.  $R^2 = 0.5950$ ), each with relatively high Adjusted  $R^2$  values (Table 7). For density modeled by  $g_3$ ,  $g_4$ , and  $g_5$ , density was significantly lower in 2011 (Year) and at control 2 shorelines compared to the reference (2010, control 1) and interactions between year and impact or control 2 were significant and positive based on parameter estimates (Table 8). Significance of both interactions indicates that both control 2 and impact shorelines differed in 2011 from the reference (control 1), so a distinct shoreline development effect cannot be inferred.

Biomass of 500- $\mu\text{m}$  infauna at Timberneck Creek responded positively to shoreline change, increasing at impact shorelines but remaining constant (Control 2) or decreasing (Control 1) at control shorelines (Figure 7B). The best-supported model for Box-Cox transformed biomass contained salinity only ( $g_7$ :  $\text{Sal}$ ;  $w_i = 0.854$ ; Adj.  $R^2 = 0.1764$ ; Table 7). Salinity was an important predictor for biomass based on parameter estimates (Table 8), and interactions between year and treatment were not included in the model and therefore not estimated.

Diversity of 500- $\mu\text{m}$  infauna at Timberneck remained constant at both impact and control 2 shorelines, but increased at control 1 shorelines (Figure 7C). However, the



best-supported model for logit-transformed diversity contained grain size only ( $g_6$ : Sed;  $w_i = 0.738$ ; Adj.  $R^2 = 0.2431$ ; Table 7). Sediment grain size was an important predictor for diversity based on parameter estimates (Table 8), and interactions between year and treatment were not included in the model and therefore not estimated. Across response variables at Timberneck for 500 $\mu$ m infauna, density was well explained by the candidate models (Adj.  $R^2 \approx 0.6$ ; Table 7), and they performed much better for 500- $\mu$ m than for 3-mm infauna (Table 5). Biomass was not well explained by the candidate models, with Adj.  $R^2 < 0.2$  (Table 7), although they performed much better for 3-mm infauna (Table 5). Finally, diversity was also not very well explained by the candidate models, with Adj.  $R^2 \approx 0.2$  (Table 7), although they performed much better for 3-mm infauna (Table 5).

Regarding community composition of 500- $\mu$ m infauna by common taxa, polychaete density increased at both control shorelines, but decreased at impact shorelines (Figure 7D). Crustaceans were largely absent from all shorelines in 2011 and they were clearly the drivers of reduced overall densities at control 1 shorelines and impact shorelines (Figure 7D). Bivalves decreased at impact shorelines but increased at control 2, with no change at control 1 (Figure 7D). Regarding species-specific community composition of 500- $\mu$ m infauna, the density of *N. succinea* increased at all control shorelines, but decreased at impact shorelines, where density was already low in 2010. The density of *C. torquata* increased at all shorelines, but more so at impact shorelines than control shorelines. Density of capitellid polychaetes increased at control 2 by a factor of about 4, but only increased slightly (factor of 1.5) at control 1, and did not increase at impact shorelines (Figure 7E). *M. balthica* drove the decrease in bivalve

density at impact shorelines, though *A. elevata* and *M. mitchelli* were also lost at impact shorelines (Figure 7E).

### *Dandy*

Density, biomass, and diversity of 3-mm infauna at Dandy did not change from 2010 to 2011 by year or shoreline treatment (Figure 8A-C). Year and salinity were correlated (VIF for Year = 40.9, Salinity = 35.9, all others < 5) at Dandy, so models containing both of these parameters ( $g_1$ : global and  $g_5$ :  $Y \times T + \text{Sal}$ ) were removed from the candidate model set for all response variables.

The best-supported models for Box-Cox transformed density of 3-mm infauna at Dandy were those containing sediment ( $g_6$ : Sed;  $w_i = 0.564$ ; Adj.  $R^2 = 0.0085$ ), salinity ( $g_7$ : Sal;  $w_i = 0.286$ ; Adj.  $R^2 = -0.0249$ ), and year and treatment ( $g_2$ :  $Y+T$ ;  $w_i = 0.135$ ; Adj.  $R^2 = 0.0098$ ), all of which performed very poorly (Table 9). None of these three models contained significant predictors for density based on parameter estimates, and interactions between year and treatment were not included in the models and therefore not estimated (Table 10).

The best-supported models for Box-Cox transformed biomass of 3-mm infauna at Dandy were those containing sediment ( $g_6$ : Sed;  $w_i = 0.560$ ; Adj.  $R^2 = -0.0082$ ) and salinity ( $g_7$ : Sal;  $w_i = 0.394$ ; Adj.  $R^2 = -0.0256$ ), and again, both of these models performed very poorly (Table 9). None of these three models contained significant predictors for biomass based on parameter estimates (Table 10).

The best-supported models for logit-transformed diversity of 3-mm infauna at Dandy were those containing year and shoreline treatment ( $g_2$ :  $Y+T$ ;  $w_i = 0.495$ ; Adj.  $R^2 = 0.1055$ ), salinity ( $g_7$ : Sal;  $w_i = 0.251$ , Adj.  $R^2 = 0.0074$ ), and sediment ( $g_6$ : Sed;  $w_i =$

0.200, Adj.  $R^2 = -0.0038$ ). Again, all models performed poorly (Table 9). None of these three models contained significant predictors for diversity based on parameter estimates, and interactions between year and treatment were not included in the models and therefore not estimated (Table 10). Across response variables at Dandy for 3-mm infauna, none of the candidate models performed well to explain any of the response variables, as all Adjusted  $R^2$  values were  $\leq 0.07$  (Table 9).

Regarding community composition of 3-mm infauna at Dandy by common taxa, there were only slight changes from 2010 to 2011 (Figure 8D). Bivalve density decreased at control 2 (marshes south of impact) and impact (riprap) shorelines, but increased slightly at control 1 (marshes north of impact) shorelines (Figure 8D). Polychaete density decreased at control 1 and decreased slightly at control 2, but increased slightly at impact shorelines (Figure 8D). In species-specific community composition of 3-mm infauna, decreases in bivalve density were due to *M. balthica* at control 2 and both *M. balthica* and *T. plebeius* at impact shorelines (Figure 8E). Decreases in polychaete density at controls 1 and 2 were driven by losses of *N. succinea* at control 1 and *C. torquata* and capitellids at control 2, whereas increased polychaete density at impact shorelines was driven by *N. succinea* and *C. torquata*. *Leitoscoloplos* spp. seemed to counter this overall trend, as they increased in density at controls 1 and 2, but decreased at impact shorelines (Figure 8E). Similarly, *D. longa* increased in density at control 2, showed little change at control 1, but decreased at impact shorelines. Spionid polychaetes recruited in 2011 to all shorelines, and seem to replace capitellids in density (Figure 8E).

Density of 500- $\mu\text{m}$  infauna at Dandy decreased from 2010 to 2011 at both control and impact shorelines (Figure 9A). Modeling output for 500- $\mu\text{m}$  infauna at Dandy should be interpreted cautiously. Low sample size made thorough interpretation of diagnostic plots difficult and also resulted in unreliable Brown-Forsythe tests for homogeneity of variance. A linear model containing year and treatment with interactions somewhat explained density (untransformed) of 500- $\mu\text{m}$  infauna at Dandy ( $g_3$ :  $Y \times T$ ; Adj.  $R^2 = 0.2378$ ; Table 11), though density did not vary significantly by year or shoreline, and the interaction between year and treatment was not significant based on parameter estimates (Table 11).

Biomass of 500- $\mu\text{m}$  infauna at Dandy increased slightly at impact shorelines but decreased at control shorelines (Figure 9B). A linear model containing year and treatment with interactions somewhat explained Box-Cox transformed biomass ( $g_3$ :  $Y \times T$ ; Adj.  $R^2 = 0.2616$ ; Table 11). Biomass also did not vary significantly by year or shoreline, but the interaction between year and treatment was significant and positive based on parameter estimates (Table 11), indicating a positive effect of shoreline modification on biomass.

Diversity of 500- $\mu\text{m}$  infauna at Dandy responded similarly to density as it decreased at both control and impact shorelines (Figure 9C). A linear model containing year and treatment with interactions somewhat explained logit-transformed diversity ( $g_3$ :  $Y \times T$ ; Adj.  $R^2 = 0.2258$ ; Table 11). Diversity also did not vary significantly by year or shoreline, and the interaction between year and treatment was not significant based on parameter estimates (Table 11). Across all three of the response variables, biomass was

predicted the best by the model, though density and diversity were predicted only slightly less well (Table 11).

Regarding community composition of 500- $\mu\text{m}$  infauna at Dandy by common taxa, bivalves decreased from 2010 to 2011 at impact shorelines but did not change at control shorelines (Figure 9D). Polychaete density decreased at all shorelines (Figure 9D). Regarding species-specific community composition of 500- $\mu\text{m}$  infauna, the decrease in bivalve density at impact shorelines was accompanied by reduced species richness from five to two species (Figure 9E), although density of *Gemma gemma* remained constant (Figure 9E). Capitellid polychaete density decreased at control shorelines by a factor of about 2, but decreased much more at impact shorelines, by a factor of about 7. Spionid polychaete density remained constant at control shorelines, but increased at impact shorelines (Figure 9E). Density of *Eteone heteropoda* decreased at control shorelines, but increased slightly at impact shorelines.

#### *Holly Cove*

Density of 3-mm infauna at Holly Cove decreased from 2010 to 2011 at all shorelines, but most sharply at control 1 (tidal wetland within Holly Cove) shorelines (Figure 10A). Year and salinity were correlated (VIF for Year = 170.3, Salinity = 174.4, all others < 5) at Holly Cove, so models containing both of these parameters ( $g_1$ : global and  $g_5$ :  $Y \times T + \text{Sal}$ ) were removed from the set of candidate models for all response variables. The best-supported models for Box-Cox transformed density were that including year and treatment with interactions and sediment grain size ( $g_4$ :  $Y \times T + \text{Sed}$ ;  $w_i = 0.573$ , Adj.  $R^2 = 0.4984$ ; Table 12) and that including sediment grain size alone ( $g_6$ :  $\text{Sed}$ ;  $w_i = 0.409$ , Adj.  $R^2 = 0.2970$ ; Table 12). For density modeled by  $g_4$ , sediment grain

size was a significant predictor, density was significantly lower in 2011 (Year) and the interaction between year and control 2 was significant based on parameter estimates (Table 13). While control 2 shorelines differed in 2011 from the reference (control 1), this was not an effect of shoreline modification. For density modeled by  $g_6$ , sediment grain size was a significant predictor and interactions between year and treatment were not included in the models and therefore not estimated (Table 13).

Biomass of 3-mm infauna at Holly Cove decreased at control 2 (tidal wetland in adjacent branch) shorelines, remained the same at control 1, and increased at impact (riprap) shorelines (Figure 10B). The best-supported models for Box-Cox transformed biomass were that containing sediment grain size ( $g_6$ : Sed;  $w_i = 0.893$ , Adj.  $R^2 = 0.3756$ , Table 12) and that containing year and treatment with interactions and sediment ( $g_4$ :  $Y \times T + \text{Sed}$ ;  $w_i = 0.106$ , Adj.  $R^2 = 0.4776$ ; Table 12). For biomass modeled by  $g_6$ , sediment grain size was a significant predictor, and interactions between year and treatment were not included in the models and therefore not estimated (Table 13). For biomass modeled by  $g_4$ , sediment grain size was a significant predictor, biomass was significantly higher at control 2 shorelines (than the reference, control 1) and the interaction between year and control 2 was significant based on parameter estimates (Table 13). While control 2 shorelines differed in 2011 from the reference (control 1), this is not an effect of shoreline modification.

Diversity of 3-mm infauna at Holly Cove remained constant at impact and control 1 shorelines, and there was a slight decline at control 2 shorelines (Figure 10C). The best-supported models for logit-transformed diversity were that containing sediment grain size only ( $g_6$ : Sed;  $w_i = 0.689$ , Adj.  $R^2 = 0.0913$ ; Table 12) and that containing year

and treatment ( $g_2$ : Y+T;  $w_i = 0.203$ , Adj.  $R^2 = 0.1159$ ; Table 12). Diversity modeled by  $g_6$  contained no significant predictors (Table 13), and for  $g_2$ , diversity was significantly higher at impact shorelines (than control 1), though interactions between year and treatment were not included in the model and therefore not estimated (Table 13). Across response variables at Dandy for 3-mm infauna, candidate models explained abundance and biomass reasonably well (Adj.  $R^2 \geq 0.3$ ), but did not explain diversity very well (Adj.  $R^2 \leq 0.2$ ; Table 12).

Regarding community composition of 3-mm infauna at Holly Cove by common taxa, bivalve density decreased at both control shorelines, but remained constant at impact shorelines from 2010 to 2011 (Figure 10D). Polychaete density decreased at control 1 and impact shorelines, but remained constant at control 2 shorelines. Chironomids appeared to recruit into control shorelines but not impact shorelines (Figure 10D). Regarding species-specific community composition of 3-mm infauna, decreased bivalve density at control shorelines was driven by decreases in *M. balthica* and *M. mitchelli* at control 1 and *T. plebeius* in addition to both *Macoma* species at control 2 (Figure 10E). Increases in bivalve density at impact shorelines were driven by increases in *M. balthica* and the addition of some *T. plebeius* and *M. lateralis*, though *M. mitchelli* decreased (Figure 10E). Thus, density of *M. balthica* decreased at both control shorelines, but increased at impact shorelines. Density of *Leitoscoloplos* spp. decreased at all shorelines in 2011 (Figure 10E). Density of *E. heteropoda* increased at all shorelines, but more so at impact shorelines than controls. Spionid polychaetes recruited in 2011 to all shorelines, and seemed to replace capitellids in density (Figure 10E).

Density of 500- $\mu\text{m}$  infauna at Holly Cove increased at impact shorelines and remained constant at control shorelines (Figure 11A). Modeling output for 500- $\mu\text{m}$  infauna at Holly Cove should be interpreted cautiously. Low sample size made thorough interpretation of diagnostic plots difficult, and also resulted in unreliable Brown-Forsythe tests for homogeneity of variance. A linear model containing year and treatment with interactions sufficiently explained density (untransformed;  $g_3$ :  $Y \times T$ ; Adj.  $R^2 = 0.7033$ ; Table 14). Density did not vary significantly by year or shoreline, but interaction between year and treatment was significant and positive based on parameter estimates (Table 14), indicating a positive effect of shoreline modification on density.

Biomass of 500- $\mu\text{m}$  infauna at Holly Cove remained constant at both control and impact shorelines, and variation among replicates was high (Figure 11B). A linear model containing year and treatment with interactions did not explain Box-Cox transformed biomass well ( $g_3$ :  $Y \times T$ ; Adj.  $R^2 = -0.3575$ ; Table 14). Biomass did not vary significantly by year or shoreline, and the interaction between year and treatment was not significant based on parameter estimates (Table 14).

Diversity of 500- $\mu\text{m}$  infauna at Holly Cove remained constant at both control and impact shorelines, and variation among replicates was high (Figure 11C). A linear model containing year and treatment with interactions did not explain logit-transformed diversity well ( $g_3$ ; Adj.  $R^2 = -0.1619$ ; Table 14). Diversity did not vary significantly by year or shoreline, and the interaction between year and treatment was not significant based on parameter estimates (Table 14). Across all three of the response variables, the model predicted density best by far (Table 14).



Regarding community composition of 500- $\mu\text{m}$  infauna at Holly Cove by common taxa, bivalve and polychaete density increased at impact compared to control shorelines, though Chironomids seemed to recruit at a greater density to control than impact shorelines (Figure 11D). Regarding species-specific community composition of 500- $\mu\text{m}$  infauna, the increase in bivalve density at impact shorelines was driven by increases in *G. gemma*, while a few *M. balthica* were lost (Figure 11E). Changes in polychaete density at impact shorelines were driven by capitellids, spionids, and *E. heteropoda*, all of which increased at impact shorelines but remained constant (capitellids) or decreased (spionids and *E. heteropoda*) at control shorelines (Figure 11E).

#### *Windy Hill*

Density of 3-mm infauna at Windy Hill decreased at impact shorelines (changed to living shoreline) and to a lesser extent at control 2 (marsh) shorelines, but remained constant at control 1 (bulkhead) shorelines from 2010 to 2011 (Figure 12A). Year and salinity were correlated (VIF for Year = 55.4, Salinity = 50.86, all others < 5) at Windy Hill, so models containing both of these parameters ( $g_1$ , global and  $g_5$ ,  $Y \times T + \text{Sal}$ ) were removed from the set of candidate models for all response variables. The best-supported models for density (untransformed) were that containing salinity only ( $g_7$ : Sal;  $w_i = 0.669$ ; Adj.  $R^2 = 0.1461$ ; Table 15) and that containing year and treatment ( $g_2$ :  $Y+T$ ;  $w_i = 0.234$ ; Adj.  $R^2 = 0.1688$ ; Table 15). For density modeled by  $g_7$ , salinity was a significant predictor (Table 16), but model  $g_2$  contained no significant predictors and interactions between year and treatment were not included in the models and therefore not estimated (Table 16).

Biomass of 3-mm infauna at Windy Hill remained constant at control 1 (bulkhead) shorelines but dropped at control 2 (marsh) and impact shorelines (Figure 12B). The best-supported models for (untransformed) biomass were that containing year and treatment with interactions and sediment ( $g_4$ :  $Y \times T + \text{Sed}$ ;  $w_i = 0.344$ ;  $\text{Adj. } R^2 = 0.3797$ ; Table 15), that containing salinity only ( $g_7$ :  $\text{Sal}$ ;  $w_i = 0.323$ ;  $\text{Adj. } R^2 = 0.2040$ ; Table 15), and that containing year and treatment with interactions ( $g_3$ :  $Y \times T$ ;  $w_i = 0.235$ ;  $\text{Adj. } R^2 = 0.3281$ ; Table 15). For biomass modeled by  $g_4$  and  $g_3$ , control 2 and impact shorelines had significantly higher biomass than reference (control 1) shorelines, and the interactions between year and impact or control 2 were significant and negative based on parameter estimates (Table 16). Significance of both interactions indicates that both control 2 and impact shorelines differed in 2011 from the reference (control 1), so a distinct shoreline development effect cannot be inferred. For density modeled by  $g_7$  there were no significant predictors (Table 16).

Diversity of 3-mm infauna at Windy Hill decreased at impact shorelines but increased at both control shorelines from 2010 to 2011 (Figure 12C). The best-supported models for 3-mm diversity were that containing year and treatment with interactions ( $g_3$ :  $Y \times T$ ;  $w_i = 0.485$ ;  $\text{Adj. } R^2 = 0.2925$ ; Table 15), that containing year and treatment only ( $g_2$ :  $Y + T$ ;  $w_i = 0.248$ ;  $\text{Adj. } R^2 = 0.1865$ ; Table 15), and that containing year and treatment with interactions and sediment grain size ( $g_4$ :  $Y \times T + \text{Sed}$ ;  $w_i = 0.180$ ;  $\text{Adj. } R^2 = 0.2950$ ; Table 15). For diversity modeled by  $g_3$  and  $g_4$ , year and shoreline did not differ from the reference, but the interaction between year and treatment was significant and negative based on parameter estimates (Table 16), indicating a negative impact of shoreline development on 3-mm infaunal diversity at Windy Hill. For diversity modeled by  $g_2$

there were no important predictors and interactions between year and treatment were not included in the models and therefore not estimated (Table 16). Across response variables at Windy Hill for 3-mm infauna, biomass was explained somewhat by the candidate models, with Adj.  $R^2$  values  $\approx$  0.2 to 0.4, though it was modeled the best of the three responses. Diversity was somewhat explained by the candidate models with Adj.  $R^2$  values between  $\approx$  0.2 and 0.3, while abundance was poorly explained by the candidate model set, with Adj.  $R^2 \leq 0.2$  (Table 15).

Regarding community composition of 3-mm infauna at Windy Hill by common taxa, polychaetes decreased at all shorelines from 2010 to 2011, but most substantially at impact shorelines (Figure 12D). Crustaceans appeared to recruit to control shorelines in 2011 where they were not found in 2010, but showed no temporal change in density at impact shorelines (Figure 12D). Regarding species-specific community composition of 3-mm infauna, the bivalve *Rangia cuneata* and spionid polychaetes increased at control shorelines, but decreased at impact shorelines (Figure 12E). Capitellid polychaetes showed a similar pattern, having increased in density at control 2 (bulkhead) shorelines and remained constant at control 1 (bulkhead) shorelines, but decreased at impact shorelines (Figure 12E). *M. balthica* clams decreased in density at both control shorelines, but changed very little at impact shorelines (Figure 12E). In contrast, *N. succinea* polychaetes decreased in density across all treatments, but most substantially at impact shorelines (Figure 12E).

Density of 500- $\mu\text{m}$  infauna at Windy Hill decreased slightly at impact shorelines and increased at control shorelines from 2010 to 2011, though variance at control shorelines is very high (Figure 13A). Modeling output of 500- $\mu\text{m}$  infauna at Windy Hill

should be interpreted cautiously. Low sample size made thorough interpretation of diagnostic plots difficult, especially given the high variability at control shorelines, and also resulted in unreliable Brown-Forsythe tests for homogeneity of variance. A linear model containing year and treatment with interactions did not explain Box-Cox transformed density well ( $g_3$ : Y×T; Adj.  $R^2 = -0.2129$ ; Table 17). Density did not vary significantly by year or shoreline, and the interaction between year and treatment was not significant based on parameter estimates (Table 17).

Biomass of 500- $\mu$ m infauna at Windy Hill remained constant at control shorelines and decreased at impact shorelines, but variance at control shorelines was again very high (Figure 13B). A linear model containing year and treatment with interactions poorly explained (untransformed) biomass ( $g_3$ : Y×T; Adj.  $R^2 = 0.1855$ ; Table 17). Biomass did not vary significantly by year or shoreline, but interaction between year and treatment was negative and significant based on parameter estimates (Table 17), indicating a negative effect of shoreline modification on biomass.

Diversity of 500- $\mu$ m infauna at Windy Hill remained constant at both impact and control shorelines (Figure 13C). A linear model containing year and treatment with interactions somewhat explained logit-transformed diversity ( $g_3$ : Y×T; Adj.  $R^2 = 0.3694$ ; Table 17). Diversity did not vary significantly by year or shoreline, and the interaction between year and treatment was not significant based on parameter estimates (Table 17). Across all three of the response variables, the model best predicted diversity, though biomass was the only response with any significant predictors (Table 17).

Regarding community composition of 500- $\mu$ m infauna at Windy Hill by common taxa, bivalve density was very low overall, with only about 60-150/m<sup>2</sup> per shoreline

treatment (Figure 13D). Polychaetes declined in density at both control and impact shorelines from 2010 to 2011, but more substantially at impact shorelines with a roughly 75% decline compared to roughly 50% at control shorelines (Figure 13D). Crustaceans increased by a factor of ~20 at control shorelines, but decreased by half at impact shorelines. Generally, responses of 500- $\mu\text{m}$  polychaetes and crustaceans seem to mirror responses of the 3-mm size class (Figure 13D). Chironomids increased in density at both control and impact shorelines, but roughly twice as much at impact shorelines as control shorelines (Figure 13D). Regarding species-specific community composition of 500- $\mu\text{m}$  infauna, the decrease in polychaetes was accompanied by a loss of *Leitoscoloplos* spp. at impact shorelines and a greater reduction of *N. succinea*, spionids and capitellids at impact than control shorelines, leaving capitellids and spionids to dominate polychaete density at both impact and control shorelines after the shoreline change (Figure 13E). The most marked change was in Crustaceans, where the density of *Corophium* spp. decreased by a factor of ~3 at impact shorelines, but increased at control shorelines by a factor of ~65. *Tanais cavolinii* was lost at impact shorelines in 2011, but did not occur at control shorelines in either year (Figure 13E).

### ***Predators***

#### *Timberneck*

Blue crab abundance at Timberneck increased at both impact 1 (boat docks) and impact 2 (riprap) shorelines, but stayed constant at both control 1 (Timberneck) and control 2 (Catlett Islands) shorelines from 2010 to 2011 (Figure 14A). A linear model containing year and treatment with interactions did not explain Box-Cox transformed blue crab abundance well at Timberneck ( $g_3: Y \times T$ ; Adj.  $R^2 = 0.0541$ ; Table 18). Blue

crab abundance did not vary significantly by year or shoreline, but the interaction between year and treatment for both impact shorelines was positive and significant based on parameter estimates (Table 18), indicating a positive effect of shoreline modification on blue crab abundance.

Abundance of fishes at Timberneck decreased at both control shorelines and impact 2, but remained constant at impact 1 shorelines (Figure 14B). A linear model containing year and treatment with interactions somewhat explained Box-Cox transformed fish abundance ( $g_3: Y \times T$ ; Adj.  $R^2 = 0.3940$ ; Table 18). Fish abundance was significantly lower in 2011 than 2010, but the interactions between year and treatment were not significant based on parameter estimates (Table 18).

Diversity of fishes at Timberneck decreased at both control shorelines and impact 1, but remained constant at control 2 shorelines (Figure 14C). A linear model containing year and treatment with interactions did not explain logit-transformed fish diversity well ( $g_3: Y \times T$ ; Adj.  $R^2 = 0.0567$ ; Table 18). Fish diversity did not vary significantly by year or shoreline, and the interactions between year and treatment were not significant based on parameter estimates (Table 18).

#### *Dandy*

Blue crab abundance at Dandy remained roughly constant at all shorelines from 2010 to 2011; standard error was large at control 1 shorelines, preventing differentiation of this mean increase other treatments (Figure 15A). A linear model containing year and treatment with interactions did not explain Box-Cox transformed blue crab abundance well ( $g_3: Y \times T$ ; Adj.  $R^2 = 0.0839$ ; Table 19). Blue crab abundance did not vary

significantly by year or shoreline, and the interactions between year and treatment were not significant based on parameter estimates (Table 19).

Abundance of fishes at Dandy decreased at both control shorelines, and abundance at impact shorelines was similar to controls in 2011, though changes in time at impact shorelines are unknown since seines were not taken before the impact took place (Figure 15B & C). Due to the lack of impact shoreline samples before development, fish abundance and diversity at Dandy were not modeled.

Diversity of fishes at Dandy remained constant at control 1 shorelines and increased at control 2 shorelines. Diversity at impact shorelines in 2011 was slightly higher than both controls (Figure 15C).

#### *Holly Cove*

Blue crab abundance at Holly Cove remained constant at control 1 shorelines, decreased at control 2 shorelines, and decreased to a higher degree at impact shorelines from 2010 to 2011 (Figure 16A). A linear model containing year and treatment with interactions adequately explained (untransformed) blue crab abundance at Holly Cove ( $g_3: Y \times T$ ; Adj.  $R^2 = 0.4254$ ; Table 20). Blue crab abundance was significantly higher at impact shorelines, but the interaction between year and impact shorelines was negative and significant based on parameter estimates (Table 20), indicating a negative effect of shoreline modification on blue crab abundance.

Abundance of fishes at Holly Cove remained constant at all shorelines, but was higher at control 1 shorelines (Figure 16B). A linear model containing year and treatment with interactions somewhat explained Box-Cox transformed fish abundance ( $g_3: Y \times T$ ; Adj.  $R^2 = 0.3395$ ; Table 20). Fish abundance did not vary significantly by year or

shoreline, but the interaction between year and control 2 was significant based on parameter estimates (Table 20). While control 2 shorelines differed in 2011 from the reference (control 1), this is not an effect of shoreline modification on fish abundance.

Diversity of fishes at Holly Cove remained constant at all shorelines, with no differences between shorelines (Figure 16C). A linear model containing year and treatment with interactions did not explain logit-transformed fish diversity well ( $g_3$ :  $Y \times T$ ; Adj.  $R^2 = -0.1919$ ; Table 20). Fish diversity did not vary significantly by year or shoreline, and the interactions between year and treatment were not significant based on parameter estimates (Table 20).

#### *Windy Hill*

Blue crab abundance at Windy Hill was very low overall, at least an order of magnitude lower than the other sites (compare Figure 17A with Figures 14A, 15A, and 16A). Blue crab abundance increased at both control sites, while showing no change at impact sites from 2010 to 2011 (Figure 17A). A linear model containing year and treatment with interactions poorly explained (untransformed) blue crab abundance at Windy Hill ( $g_3$ :  $Y \times T$ ; Adj.  $R^2 = 0.1943$ ; Table 21). Blue crab abundance was significantly higher in 2011 than 2010, and the interactions between year and impact or control 2 were significant and negative based on parameter estimates (Table 21). Significance of both interactions indicates that both control 2 and impact shorelines differed in 2011 from the reference (control 1), so a distinct shoreline development effect cannot be inferred.

Abundance of fishes at Windy Hill remained constant at control 1 shorelines, but was higher at control 2 and impact shorelines during both 2010 and 2011, though there is



a high degree of variability at both control 2 and impact shorelines in 2010 (Figure 17B). A linear model containing year and treatment with interactions adequately explained Box-Cox transformed fish abundance ( $g_3$ :  $Y \times T$ ; Adj.  $R^2 = 0.41319$ ; Table 21). Fish abundance did not vary significantly by year or shoreline, and the interactions between year and treatment were not significant based on parameter estimates (Table 21).

Diversity of fishes at Windy Hill increased at both control 2 and impact shorelines, but decreased at control 1, though variability was high at all shorelines (Figure 17C). A linear model containing year and treatment with interactions somewhat explained logit-transformed fish diversity ( $g_3$ :  $Y \times T$ ; Adj.  $R^2 = 0.2073$ ; Table 21). Fish diversity did not vary significantly by year or shoreline, but the interactions between year and impact or control 2 were significant and positive based on parameter estimates (Table 21). Significance of both interactions indicates that both control 2 and impact shorelines differed in 2011 from the reference (control 1), so a distinct shoreline development effect cannot be inferred.

### *Summary*

Responses to shoreline modification were variable by site and organism. Infaunal density and biomass responded positively to shoreline modification at Timberneck, Dandy, and Holly Cove, but negatively at Windy Hill, whereas infaunal diversity responded negatively at Timberneck and Windy Hill (Table 22). Blue crab abundance responded positively at Timberneck, did not change at Dandy, and responded negatively at Holly Cove (Table 22). Blue crab abundance decreased at Windy Hill, though a distinct shoreline effect cannot be inferred (Table 22). Fish abundance and diversity showed no distinct shoreline response at any site (Table 22).

## DISCUSSION

Using a before-after, control-impact study aimed at examining effects of shoreline modification, changes in benthic infauna and predators were detected, sometimes attributable to the shoreline modification, or otherwise attributable to secondary environmental effects that changed in accordance with shoreline modifications.

### *Effects of Shoreline Modification on Infauna*

Overall, 3-mm infauna tended to show significant responses for biomass and/or diversity at shoreline modification sites (seen in AIC models with significant treatment by year interactions), with large-scale shoreline impacts (> 80 m), though the direction of change was sometimes opposite of that hypothesized. Timberneck Creek (shoreline changed from natural to riprap and boat docks) showed positive changes in biomass but negative changes in diversity, but Windy Hill (bulkhead to living shoreline) showed negative changes in diversity. Responses of 3-mm infauna were often related to changes sediment grain size or physical structure accompanying shoreline modifications, leading to either opportunistic species or species common to the new physical conditions (*e.g.*, sediment type).

Responses of 500- $\mu$ m infauna to shoreline modification are particularly noteworthy, as effects of disturbances are more likely to manifest in smaller individuals and recruits. Larger species and adults (represented in 3-mm samples) can react differently than small species and recruits (represented in 500- $\mu$ m samples) to disturbance, as seen in a previous BACI study where recruitment of *Cerastoderma edule* and *Macoma balthica* decreased for up to 8 years following dredge disturbance, whereas adults were largely unaffected (Piersma et al. 2001). Juveniles and smaller organisms

with shorter-life spans are likely to respond before changes become evident for larger species or adult populations, which are more likely to reflect longer-term changes (Levin 1984, Piersma et al. 2001). For example, the Dandy and Holly Cove, which were subject to shorter lengths of developed shoreline, showed significant effects of shoreline development on 500- $\mu$ m infauna but not on the 3-mm infauna. At Timberneck Creek, significant impacts were inferred from shoreline development only for 3-mm infauna, but may have influenced 500- $\mu$ m infauna as well. Density of 500- $\mu$ m infauna at control 2 sites (at Catlett Islands, an adjacent but similar system) deviated significantly from control 1 sites (within Timberneck Creek), which more closely resembled impact sites than control 2 sites. This suggests a potential spillover of shoreline disturbance for 500- $\mu$ m infauna, meaning that shoreline development may have disturbed recruitment of benthic infauna throughout Timberneck Creek, affecting both impact and control 1 sites.

#### ***Effects of Shoreline Modification on Predators***

Shoreline development resulted in a significant increase in blue crab abundance at both impact sites at Timberneck. This likely occurred because blue crabs feed on benthic organisms (Virnstein 1977, Seitz et al. 2001), tend to have smaller foraging areas and may be more heavily reliant upon the benthos at the particular shorelines than fishes (Clark et al. 1999, Hines et al. 2009). At Timberneck, biomass of 3-mm infauna increased significantly at impact 1 (boat docks) potentially allowing for more food for blue crabs at impact sites (*i.e.*, bottom-up control). Blue crabs aggregate around structure (Lipcius et al. 2007), and the inclusion of boat docks, pilings, and riprap at Timberneck may represent potential sources of structure resulting in aggregation of crabs rather than a true increase in their abundance.

In contrast, at Holly Cove, blue crab abundance responded negatively to shoreline modification. Though 500- $\mu\text{m}$  density increased, the small infauna that increased in density may not be preferred prey items for blue crabs (Lipcius et al. 2007).

Blue crab abundance at Windy Hill was characterized by many tows with zero abundance, particularly at impacted shorelines. This may be a result of sampling bias. For 2010 samples, there were not many blue crabs collected, and those collected were almost always large adult males, which was not the case at the other study sites, where juvenile crabs were common in scrape samples. It is possible that the crab scrape method led to inefficient sampling and bias for 2 reasons: (1) sampling was not able to capture crabs given the more sparse distribution of blue crabs in Maryland than Virginia (King et al. 2005), and (2) the crab scrape could not be set within the small coves of the living shoreline in 2011 where crabs, especially juveniles, are more likely to be found associated with structure (Lipcius et al. 2007).

Fish abundance and diversity showed no significant effects of shoreline modification at any site, which may have been a result of high variability, as fishes are more transient with exception of directly shoreline-associated species such as mummichogs, which are not benthic feeders (Kneib 1984; Minello et al. 2003). The lack of shoreline modification influence on fishes indicates (1) a general lack of impact of shoreline modification at the scales studied on fishes, or (2) the need for either more sampling or more efficient fish sampling strategies. The need for more efficient sampling is particularly evident at shorelines without a beach, which is a common characteristic of impacted shorelines in this study, as it is much more difficult to ensure an efficient seine

was taken without a beach (Williams 2010). A different sampling method, such as a fyke net (Fago 1998) or minnow traps might be a more successful alternative approach.

At Windy Hill, seines were particularly difficult to pull through the impact shorelines, as logs and trees were included in the coves to provide habitat structure, which may have introduced sampling bias. Additionally, while seine abundances were corrected for area, different techniques and collection equipment used by the two labs over the different years may have influenced abundances.

Diversity is not as sensitive to differences in sampling techniques as abundance as it depends on ratios of species abundance to richness, may have been subject to less sampling bias. Impact and marsh control shorelines at Windy Hill mirrored one another, and diversity increased at these sites from 2010 to 2011. Fish species are more motile and may be faster responders to positive changes in shoreline condition than infaunal benthos, providing a first look at the positive effects of living shoreline restoration.

### ***Environmental Drivers***

Several models included sediment type (Timberneck 3-mm infaunal biomass and diversity, Timberneck 500- $\mu$ m infaunal diversity, Holly Cove 3-mm infaunal density and biomass) and salinity (Timberneck 500- $\mu$ m infaunal biomass, Windy Hill 3-mm infaunal density) as significant predictors. For Timberneck 3-mm biomass and diversity, both the year and impact shoreline interaction and sediment were important predictors, indicating that sediment likely changed with shoreline modification. This suggests the importance of monitoring sediment changes with shoreline modification, especially as biomass and diversity responded differently to changes in sediment grain size. At Timberneck impact 2 shorelines (tidal wetland to riprap), infaunal community-wide metrics (density, biomass,

and diversity) did not change significantly but species composition did. The polychaete *Clymenella torquata* was largely responsible for the increase in density at impact 2, though their density did not change much at other sites. In 500- $\mu$ m infauna at impact 2, *C. torquata* increased and *M. balthica* decreased, driving the overall decrease in bivalve density. This corresponds with the large increase in % sand at impact 2 (riprap) after development, as *C. torquata* inhabits muddy sand in high densities (Lippson and Lippson 2006), and *M. balthica* prefers muddier environments (Seitz et al. 2001). The change in sediment grain size and subsequent changes in community composition suggest that faunal changes resulted from changes in habitat.

At Holly Cove, sediment was an important predictor for 3-mm density and biomass, so changes in sediment grain size between years may have also influenced 500- $\mu$ m density where infaunal changes with shoreline modification were detected, but models with sediment were not compared for the smaller size class due to small number of samples processed.

Shoreline modification at Windy Hill resulted in an observable change in bottom type and the largest increase in fine sediments across all sites. The impact site (a bulkhead shoreline prior to modification) was characterized by predominantly gravel and sand with a little bit of clay, while after one year, about 10 cm of silty and detrital material had collected in the constructed coves. Although grain size was not a significant predictor of 3-mm infauna, grain size measurements may not have been optimized for this site as the current grain size protocol does not include a specific method to remove detritus (Plumb 1981), and detrital material ended up in coarse sediment fractions, where it may have prevented detection of sediment changes at Windy Hill impact shorelines.

For future studies, removing detrital material by elutriation may help to improve sediment grain size classification at sites with high levels of detritus.

Given the observed changes in sediment grain size, an examination of sediment organic content would be helpful to determine whether changes in sediment grain size also translate to changes in organic input, which may alter biomass with more available food for production (Pearson and Rosenberg 1978). Samples were taken for sediment TOC / TN analysis, but at this writing have not yet been analyzed.

Timberneck 500- $\mu\text{m}$  infaunal biomass and Windy Hill 3-mm infaunal density both decreased as salinity decreased, suggesting the importance of monitoring salinity in this study. In 2010, mean river discharge into Chesapeake Bay was 51.6 gallons per day, whereas in 2011 it was 73 gallons per day (Chesapeake Bay Program 2012), leading to a decrease in salinity during the course of the current study. Macrobenthic abundance is known to show systematic changes with long-term physicochemical environmental changes, particularly in response to salinity gradients, and increases in productivity with increased salinity (Möller *et al.* 1985; Diaz and Schaffner 1990). In a 14-year study of mesohaline mid-Chesapeake Bay, annual freshwater inflow decreased, salinity increased, and bottom DO below the pycnocline declined (Holland *et al.* 1987). Concomitantly, eurytolerant marine species, and particularly ubiquitous opportunists responded to increased salinity by increasing their abundances, whereas estuarine species declined. The four current study sites are all mesohaline; Dandy is high mesohaline, Timberneck and Holly Cove mid-mesohaline, and Windy Hill is low mesohaline, so their responses should be considered within their large-scale settings.

Environmental drivers are important for managers to consider when modifying shorelines. If sediments are kept similar to the original condition, species composition is more likely to remain largely unchanged, but species composition changes with sediment grain size. If salinity changes in an area, near-shore communities are likely to respond regardless of changes in the local habitat.

### *Scope of the Present Study Versus Previous Studies*

While the results of this study did not directly support the original hypotheses, they did complement findings of previous studies by providing direct evidence of the ability of shoreline modification to influence near-shore community structure. To explain this discrepancy, it is important that the current study and previous studies from which hypotheses were developed be examined within their experimental scope.

Variability in density, biomass, and diversity of the benthos is very high given the patchy nature of benthic infauna (Holland 1985, Dauer et al. 1987, Diaz and Schaffner 1990, Snelgrove and Butman 1994) and may have been higher than that captured conclusively by the sampling effort. Specifically, for 500- $\mu\text{m}$  infauna at Dandy, Holly Cove, and Windy Hill, sample sizes of analyzed data were low, making the interpretation of results difficult. Spionid polychaetes were observed to change in density with shoreline development at both Dandy and Holly Cove, and this consistency in results is encouraging, but given the high variability in biomass and low adjusted  $R^2$  values at Dandy, these results should still be interpreted with caution. Density of 500- $\mu\text{m}$  infauna at Holly Cove increased at impact shorelines accompanied by low variability and a relatively high adjusted  $R^2$ , which provide more compelling evidence of a significant



effect of shoreline modification despite the small sample size. Increasing the sample size for 500- $\mu\text{m}$  infauna will be helpful to further elucidate shoreline development effects.

Additional stressors outside the scope of the study may have been acting on near-shore communities and driving community changes in addition to shoreline changes. Recent work suggests shoreline modification interacts with upland use and percent of developed shoreline within a system (King et al. 2005, Bilkovic and Roggero 2008, Seitz and Lawless 2008, R.D. Seitz *unpublished data*) Blue crab abundance varies with not only shoreline condition (as % of marsh along 100 m of shoreline) but with watershed land use and salinity (King et al. 2005). Nekton assemblages at natural or riprap shorelines in watersheds with < 23% upland development are different from those at bulkhead shorelines (Bilkovic and Roggero 2008). In the lower Chesapeake Bay, diversity and abundance of subtidal benthic infauna is higher adjacent to natural marsh than to bulkhead shorelines; however, abundance and diversity are high (low) adjacent to riprap if shorelines throughout the system are (not) mostly undeveloped (Seitz and Lawless 2008). Thus, it is important to consider upland development and the percent of developed shoreline (scope for change) with the effects of shoreline development.

The length and age of impacted shorelines may have impacted the detection of shoreline modification effects in this study. Previous studies indicate decreases in near-shore community metrics (density, biomass, and diversity) at developed as compared to natural shorelines compared stretches of impact shoreline that are generally longer than those in the present study (> 50 m Seitz et al 2006, > 80 m Bradley 2011), and at shorelines that have been developed for much longer than those in this study (*e.g.* sites selected from shoreline maps developed 3 years prior to sampling; Bradley 2011).

Modified shoreline lengths at Dandy (49 m), Holly Cove (34 m), and Timberneck (84, 81, 37, and 26 m, as discontinuous sections) tended to be shorter than those in previous studies, and may have been small enough and surrounded by source populations.

Shoreline development and blue crab abundance are negatively correlated in Chesapeake Bay, but at shoreline stretches at least 100 m long (King et al. 2005). In Puget Sound, WA, abundances of insects and talltrid amphipods is reduced on shorelines altered with bulkhead or riprap compared to beach at shorelines over 100 m in length (natural shorelines), and up to 200 m (altered shorelines; Sobocinski et al. 2010, K. Sobocinski *pers. comm.* 2012). In contrast, the present study was designed to examine direct effects of shoreline modification disturbances before and after modification rather than compare communities at static shoreline habitats. Interestingly, Windy Hill, where the impacted shoreline was longest (149 m of bulkhead removed and replaced with 309 m of shoreline around breakwaters), was the only site to show negative responses of infaunal communities to shoreline modification. While the decrease in biomass and diversity at Windy Hill contrasts the original hypotheses, the overall scope of the shoreline change was much greater at Windy Hill than the other three study sites, which may have led to a large disturbance effect that was still apparent after one year. Windy Hill's shoreline modification altered the shape of the shoreline, its linear length, and resulted in changes farther channel-ward than the other sites.

Results of many previous studies of shoreline development effects that found decreased community metrics adjacent to developed shorelines did not directly examine changes before and after modification or with ecological succession, but generally examined differences at shoreline types as static habitats. However, previous studies

with similar designs to this study that included repeated sampling tended to concur with the results of the present study. In a study that monitored benthic assemblages during and after construction of a bulkhead in highly polluted Kyeonggi Bay, Korea, there were significant over two-fold increases in total infaunal community biomass and abundance and significant increases in sediment grain size after completion of the bulkhead compared to samples taken during its construction (Ahn and Choi 1998). Species diversity did not change, though species-specific changes favored organisms that live in coarser sediments, such as the paraonid polychaete *Aricidea* spp. and orbiinid polychaete *Scoloplos armiger* (Ahn and Choi 1998). In addition, the predominance of K-selected species was reduced, indicating that the new communities are probably undergoing ecological succession after the disturbance. The findings of Ahn and Choi (1998) are similar to those at my Timberneck, Dandy, and Holly Cove sites, in that grain size tended to increase, and infaunal abundance and biomass tended to increase driven by opportunistic species. While the Ahn and Choi (1998) study is on a much larger scale than the present study, its more similar sampling structure than most previous studies and more similar findings emphasize the importance of studying communities through time to characterize the disturbance caused by developing shorelines.

In a before-after control-impact (BACI) study of a bulkhead site modified with installation of a living shoreline, densities of some marsh-associated organisms increased after just two months, indicating that some organisms can respond immediately to restoration (Davis et al. 2006), and emphasizing the need to look at individual species-level responses. Additionally, in a BACI study where recruitment of *C. edule* and *M. balthica* were studied following dredge disturbance, adults and juveniles respond

differently, emphasizing the need to examine different ages and size classes of organisms separately (Piersma et al. 2001).

### ***Opportunistic Species and Habitat Changes Drive Significant Impacts of Shoreline Modifications***

Anthropogenic environmental disturbances, such as habitat destruction (Thrush et al. 2001), hypoxia (Dauer 1993), toxic pollution (Gray 1979, Dauer 1993, Preston and Shackelford 2002), and dredge spoil disposal (Rhoads et al. 1978) reduce biological diversity and/or biomass of soft-bottom infauna (Dauer 1993, Thrush et al. 2001, Preston and Shackelford 2002), and their effects on in time and with ecological succession have been documented. Biomass and diversity can be reduced in favor of opportunistic species that respond quickly after a disturbance, though the density of these organisms can increase rapidly and be highly variable (Dauer 1993, Thrush et al. 2001).

Opportunistic species in marine systems respond by rapidly increasing their population densities immediately after disturbances, such as pollution (Pearson and Rosenberg 1978), hypoxia (Santos and Simon 1980, Dauer 1993), dredge spoil disposal (Rhoads et al. 1978, Newell et al. 1999), and sediment movement (Zajac and Whitlatch 1982, Commito 1995).

Within estuarine environments, reference benthic infaunal communities contain species with combinations of both r-selected and K-selected traits (Schaffner 1990, Newell et al. 1998, Schaffner 2010), including many opportunistic species as characterized by their life histories and responses to disturbance (Boesch 1977, Holland et al. 1987, Marsh and Tenore 1990, Schaffner 1990). Opportunists tend to have short life-spans, rapid reproduction in favorable habitats, and tolerance to a wide range of

environmental conditions, and are often adapted to feed at the sediment-water interface and brood early-development stages (Santos and Simon 1980, Holland et al. 1987).

Infaunal communities in Chesapeake Bay are composed largely of species considered to be euryhaline opportunists, and are conspicuous in macrofaunal assemblages (Boesch 1977, Diaz 1984, Holland et al. 1987, Marsh and Tenore 1990). Within Chesapeake Bay, euryhaline opportunistic species include spionid polychaetes (*Streblospio benedicti* and *Polydora ligni*; Rhoades et al. 1978, Santos and Simon 1980, Diaz 1984, Marsh and Tenore 1990), clam worms (*Neanthes succinea*; Holland et al. 1987), small bivalves (*Gemma gemma*, *Mulinia lateralis* and *Macoma mitchelli*; Rhoades et al. 1978, Santos and Simon 1980, Holland et al. 1987, Commito et al. 1995, Schaffner 2010), and amphipods (e.g. *Leptocheirus plumulosus*, which is not euryhaline but an estuarine endemic opportunist; Holland et al. 1987, Marsh and Tenore 1990).

In the present study, opportunistic species drove significant responses in community metrics to shoreline modification at three of four sites. At Dandy, the positive effect on 500- $\mu$ m biomass was driven by an increase of spionids and maintenance of *G. gemma* while all other (larger) bivalves decreased. At Holly Cove, the positive effect on 500- $\mu$ m density was also driven by an increase of both spionids and *G. gemma*. At Windy Hill, the negative effect on 3-mm diversity was driven by a loss of spionids and *N. succinea*, whereas density of *M. balthica* remained largely the same. The negative effect on 500- $\mu$ m biomass at Windy Hill was driven by the recruitment of corophiid amphipods at control shorelines but not impact shorelines, and a reduction in *N. succinea* and spionids at all shorelines, but most severely at impact shorelines.

Within mesohaline and polyhaline Chesapeake Bay, annual recruitment of infauna classified as opportunists follows a single sharp increase in density once per year, in spring and early summer (in contrast to their continued recruitment in other locations), following the spring bloom in water column production when sedimentation rates of organic matter are high (Holland et al. 1987, Baird and Ulanowicz 1989, Marsh and Tenore 1990). This peak is quickly followed by a rapid decrease, resulting in a single boom and bust for opportunistic species (Holland et al. 1987). The spionid polychaete *S. benedicti* (Diaz 1984, Marsh and Tenore 1990) and the amphipod *L. plumulosus* (Holland et al. 1987, Marsh and Tenore 1990) exhibit very large recruitment in spring (mesohaline, Marsh and Tenore 1990) or summer (polyhaline, Diaz 1984), with the increase in labile organic matter available for deposit feeding, followed by mass mortality after one or two months after depletion of the organic material. The small bivalve *G. gemma*, is a eurytolerant marine species capable of rapidly increasing its population size in patchy, favorable environments as it reproduces with a short generation time, broods its young, and disperses by passive bedload transport (Commuto et al. 1995). High densities of *G. gemma* provide evidence for a disturbance effect, and *G. gemma* has been shown at high density adjacent to riprap (Bradley 2011). Alternatively, longer-lived, larger species such as *M. balthica* show an increase in abundance in late spring or summer followed by slower growth stages and an increase in standing stock biomass for (Holland et al. 1987). Larger infauna may take longer to respond and are also subject to changes in abundance with longer-term physiochemical environmental changes (*e.g.* salinity, sediment changes, and hydrodynamics; Schaffner 2010).

While opportunistic species are expected to increase around or soon after the time of sampling, the BACI design allows for the separation of an increase in opportunists due to their natural boom and bust dynamics from an impact of shoreline modification. A normal temporal increase in response to the spring bloom would have been seen at all shorelines, not just at impacted shorelines. Furthermore, if infauna were simply responding to large-scale changes in their environment, such as the decrease in salinity found this study from 2010 to 2011, a decrease in euryhaline marine organisms and opportunists would be expected. At Dandy and Holly Cove, this did not happen; eurytolerant opportunists increased in spite of decreased salinity, and their response is seen well after the typical opportunistic response time frame, supporting the notion of a prolonged response to shoreline development. At Windy Hill, eurytolerant opportunists declined, but estuarine endemics, such as *M. balthica* did not increase, suggesting an additional driver of change besides salinity alone. In addition, a typical initial opportunistic species' response to a pulse disturbance within Chesapeake Bay occurs within a short timeframe such as weeks or months (Diaz 1994, Schaffner 2010) as compared to the full year between sampling events in this study.

Given that opportunistic species drove community responses to shoreline modification, but opportunists did not respond at control shorelines, the timeline of change does not fit the classic opportunistic response timeframe, and infaunal responses were not driven by large-scale changes in salinity, shoreline modification at Dandy, Holly Cove, and Windy Hill may represent a disturbance to these systems and thus a fundamental change in habitat. With continued responses of infauna one year after the disturbance, there have likely been fundamental changes at the altered shorelines that

either favor opportunists (Dandy and Holly Cove) or restrict them (Windy Hill).

Potential changes to habitats at impacted shorelines include sedimentary changes (as evidenced by grain size changes at Holly Cove and Windy Hill), enhanced wave energy due to hardening, or altered food availability.

In contrast, changes in infaunal communities at Timberneck with shoreline modification were not driven by opportunistic species. At Timberneck impact 1 (boat docks), 3-mm infauna experienced a positive effect on biomass driven by an increase in capitellid polychaetes and a negative effect on diversity driven by increases in *M. balthica* and capitellids as dominant species. Capitellid polychaetes include some opportunistic species, such as *Capitella capitata*, but the most common capitellids in shallow mesohaline Chesapeake Bay are *Mediomastus ambiseta*, and *Heteromastus filiformis*. *M. ambiseta* exhibits relatively prolonged recruitment and low mortality (Diaz 1984, Holland et al. 1987) and is sensitive to pollution (Weisberg et al. 1997), thus it is not an opportunist. *H. filiformis* is larger, long-lived, and eurytolerant, but with its relatively large standing stock biomass, it is not considered to be opportunistic (Holland et al. 1987). Without species-level identifications, it is difficult to determine if the response at Timberneck was driven by opportunistic species, though the changes at Timberneck are also indicative of an alteration in habitat that has lasting impacts on resident infauna, concordant with observed changes in sediment grain size at this site.

Examination of community structure over a longer period after shoreline changes would be helpful to monitor any further changes in habitat and characterize continued responses to habitat changes caused by shoreline modification.

### ***Characterizing Shoreline Modification as a Stressor on Estuarine Benthic Infauna***



Ecological stressors are chemical or physical perturbation events that alter the environment and may or may not lead to a response in a biological system- such as an organism, population, or community of interest, whereas ecological disturbances are perturbation events caused by humans, and include anthropogenic stressors (adapted from White and Pickett 1985, Underwood 1989, and MarLIN Glossary 2005, see Constable 1999 and Elliott et al. 2007 for reviews). Ecological resistance is the degree to which a variable resists change in structure or behavior in the face of disturbance (Holling 1986) or is changed following a perturbation (Costanza et al. 1992). Resilience is the ability of an ecosystem to return to its original state after being disturbed (MarLIN Glossary 2005).

Chesapeake Bay is a semi-enclosed drowned-river valley estuary where ocean water is mixed and diluted with freshwater, and as such is subject to fluctuations in sedimentation (coarse sands and fine muds), water and air temperature, freshwater delivery, seasonal stratification and mixing, nutrient delivery, and an intense diatom-based spring bloom, making it a highly variable environment. Given their immobility in a highly variable environment, resident infauna in Chesapeake Bay are naturally resilient to stressors, partially due to the abundance of opportunistic species (Elliott et al. 2007). In addition, natural and anthropogenic stressors are common in estuaries, and occur within a high degree of spatial and temporal variability (Dauer et al. 1993, Newell et al. 1998, Elliott et al. 2007), preventing estuarine infaunal communities from reaching ecological "equilibrium". In lieu of equilibrium communities, managers are more interested in measuring recovery back to a reference condition, such as that found in communities at similar but undisturbed habitats (Weisberg et al. 1997, Schaffner 2010).

In the lower York River Estuary, a major mesohaline tributary to Chesapeake Bay, sediments are “biologically dominated” in that they are constantly being reworked by infaunal bioturbators and bioirrigators, while those in the upper estuary are "physically dominated" and subject to high variability of physical drivers such as salinity and sedimentation (Schaffner et al. 2001, Dickhudt et al. 2009). Constant bioturbation or physical variability represent a constant state of flux within the benthos, making its residents tolerant and stable to disturbance, and potentially leading to lesser influences of shoreline disturbances on an already highly dynamic infaunal community. Given the highly dynamic setting, observation of effects of shoreline modification within estuaries such as Chesapeake Bay provides compelling evidence of the ability of shoreline modification to impact near-shore communities.

Shoreline modifications have the capacity to act as pulse or press disturbances depending on the environment in which they act. Recovery from short-term (pulse) disturbances, such as organic enrichment, pollution, changes in water quality, depends on the size of the patch disturbed, the species available to colonize the patch (as larvae or adults Levin 1984), and species interactions (Zajac and Whitlatch 1982). Recovery from these long-term (press) perturbations, such as dam placement, or climate change (Lake 2000, Schiel et al. 2004) depends on the organisms available to recolonize the patch, the life histories and dispersal mechanisms of these organisms, spatial and temporal variability of the stressor, and interactions between the stressor and other stressors or environmental factors (Constable 1999).

At Windy Hill, shoreline modification was the most extensive in length and distance of influence channel-ward of the study sites examined. Results after several

years may reflect an improvement in habitat quality from the addition of a living shoreline, represented by a press disturbance. The two controls at Windy Hill were designed to represent the beginning (bulkhead) and ending (marsh) shoreline conditions. In time, as impact shorelines more closely mirror marsh shorelines than bulkhead shorelines, this press disturbance resulting in a permanent increase in community metrics will be realized. Currently, density and biomass of 3-mm infauna as well as crab abundance and fish abundance and diversity at impact sites more closely followed trends at marsh (control 2) shorelines.

Additionally, the density of deep-dwelling *M. balthica* clams decreased only slightly whereas most organisms decreased more markedly, which may reflect the adaptive ability of *M. balthica* in several ways: (1) adult *M. balthica* clams are deep burrowers (Hines and Comtois 1985) and may be buried deep enough in the sediment to be protected from major physical damage during construction, (2) *M. balthica* adapts to changes in sediment grain size and resuspension of sediments by performing both filter and deposit feeding (Lin and Hines 1994), and 3) fine sediments which contain the preferred food source of *M. balthica* increased at impact shorelines (Lin and Hines 1994). Infaunal samples taken two years post-construction at the impact shoreline show a marked increase in abundance and diversity of large clams (R.D. Seitz, *unpublished data*), and are likely indicators that the living shoreline modification takes longer than one year for benefits to the benthos to manifest.

While the other shoreline modifications in the present study had shorter continuous stretches of impact shorelines, the changes in the infaunal communities at these sites indicated that shoreline modification is likely to manifest as a press

disturbance even in otherwise healthy environments, if they have undergone a change in habitat such as in sediment grain size or organic matter (Underwood 1994). Shoreline modifications at Timberneck, Dandy, and Holly Cove sites in this study are likely to represent press impacts, as density and biomass of infaunal communities increased as a result of an influx of opportunistic species that continue to respond, despite the fact that these sites are not subjected to a high degree of external stressors, adjacent to otherwise largely pristine environments, and are likely to have ample resources (*e.g.* recruits, allochthonous carbon). In addition, shoreline modifications that impact long stretches of shoreline, extend far channel-ward, or are part of environments with additional stressors are also likely to represent a (more intense) press impact, particularly when they sever long stretches of the land-water interface, removing sources of allochthonous carbon from detrital marsh material, or induce additional stressors (such as altered hydrodynamics, increased runoff from upland development). Shoreline modification at Windy Hill is likely to represent more intense press impact than the other sites given its sediment disturbance and alterations of hydrodynamics and reintroduction of the land-water interface to an environment that was degraded prior to the change to living shoreline (in 2010, biomass and diversity were lower here than at any of the other sites), evidenced by the lack of community improvement after one year. In opposition to the other sites, however, sediment may stabilize with maturity of the living shoreline at Windy Hill, and the press impact is expected to result in positive community responses as the bulkhead has been removed, reducing wave exposure in the created protected embayments, and restoring the land-water interface and potentially increasing allochthonous carbon inputs. In otherwise relatively pristine environments, where habitat

conditions are unchanged, shoreline modifications may manifest as pulse disturbances, but this result was not seen in the present study.

Given the characteristics of benthic infauna in Chesapeake Bay, the importance of opportunistic species in driving responses to shoreline modification, and their continued response after a year, it is reasonable to assume that recovery from the shoreline disturbances at Dandy, Holly Cove, and Windy Hill are not complete, and the communities have not returned to their reference conditions.

### ***Suitability of the BACI Design and Linear Modeling***

The BACI study design is particularly well suited to meet the need of examining effects of shoreline modification on community succession, as it provides direct evidence of shoreline modification impacts by comparing communities undergoing change to controls both before and alongside the change in time, controlling for age of the shoreline change, and allowing for the comparison of multiple years of study.

In addition, most studies that monitored artificial habitats and shoreline modifications examined patterns- what species are present, in what assemblages and associations. Little research effort has taken on an examination of the process of changes due to shoreline modification (Chapman and Underwood 2011), which is a significant advancement afforded by the BACI study. For instance, if early successional species arrive after shoreline modification at the peak of predation, they may be highly susceptible to predation and unable to recolonize a healthy assemblage. Knowledge of this process may allow for better advising as to when to complete projects to achieve the best possible assemblage of infauna.

Expanding on the standard BACI-ANOVA approach, the present study employed the use of linear models with additional predictor variables to help attribute some of the variability in benthic infaunal responses to known benthic estuarine predictors, salinity and sediment grain size. This study illustrates the value of including environmental drivers besides the driver of interest (shoreline modification) in various combinations of models to be compared. In several cases, sediment grain size or salinity was more important in modeling community responses than shoreline treatments, stressing the importance of outside environmental variables in modeling benthic communities. Comparing linear models within the BACI sampling framework is also well suited to examine community responses to multiple stressors such as upland use and shoreline modification, and look for interactions between these stressors.

With previous work characterizing succession and community response after disturbances such as nutrient enrichment (oil spills, sewage drainage, pulp mill wastes; Pearson and Rosenberg 1978) and disposal of dredged material (Rhoads et al. 1978, Newell et al. 1998, Schaffner 2010) and the use of this information in characterizing the quality of the benthos for management purposes, consideration of infaunal benthic community succession after different types (length, complexity) of shoreline modification may help managers make decisions about the use of different practices and develop indicators for management use.

To date, examination of shoreline development effects through time has been limited and has very rarely been examined with a before-after-control-impact study. Previous studies have examined communities during and after development with bulkhead, but lacked control sites (Ahn and Choi 1999), or examined bulkheads (Bulleri

et al. 2005, Sobocinski et al. 2010) and riprap (Sobocinski et al. 2010) through time with controls, but without sampling prior to shoreline change. The importance of the present study is largely driven by its inclusion of samples before and after development, which allows simultaneous examination of control (unchanged) and impacted shorelines, as well as control over the age of the shoreline, as succession of communities back to their reference conditions may occur over different time frames in different environments. For instance, upper Chesapeake Bay infaunal communities in muddy sand environments take 18 months to recover from dredge disturbance (Pfitzenmeyer 1970), whereas freshwater James River infaunal communities in semi-liquid muds take only about 3 weeks to recover from dredge disturbance (Diaz 1994). Thus, macrobenthic succession occurs over different time scales after dredge disturbance for different habitats (Newell et al. 1998), emphasizing the need for monitoring changes after shoreline modification in time in different habitats. Adults and longer-lived species may take longer to recover than the single year included in this study, whereas recruits may show responses in much shorter time frames (Levin 1984), and may have already rebounded between the two years in the present study, especially if shoreline modification represents a pulse disturbance.

Additional years of study would be beneficial to this study especially in the case of Windy Hill, where recovery and acclimation of the community to the living shoreline is expected to take some time. For instance, there may be a large influence of modification at first, particularly on opportunistic species (a pulse impact or disturbance) followed by a rebound in the community, but as time passes, the severance of the land and water interface may become more influential, with a long-term consequence on the community (a press impact). If responses to shoreline modification change over time,

ages of shorelines should be accounted for in future studies (*i.e.* mature marsh shorelines are not comparable to new bulkhead shorelines, old bulkhead shorelines are not comparable to new riprap shorelines, *etc.*). With continued sampling, it may be possible to capture the longer-term responses to shoreline modification and determine the major species that influence the response as ecological succession continues. This information would help inform management of coastal systems.

## CONCLUSIONS

In spite of the variability of trends with shoreline modification, and the differing direction of responses as compared to previous studies, the present study was able to corroborate previous findings in that shoreline modification impacted near-shore communities. Impacts of shoreline modifications on infauna were predominantly driven by increases in opportunistic species, which suggests the importance of viewing the results in light of ecological succession after a disturbance. Impacts of shoreline modifications on blue crabs varied by site, with increased blue crab abundance adjacent to added structure (Timberneck) and decreased blue crab abundance with smaller, potentially less-preferred prey species (Holly Cove). The lack of impacts on fishes indicates their motility and larger foraging range.

Results from the present study in opposition to the direction of those in previous literature are likely a result of major differences between this and previous study designs, most notably (1) the inclusion of small-scale shoreline modifications in the present study, (2) the lack of information regarding age and degree of succession of altered shorelines in



previous studies, and (3) the influence of succession after a disturbance in the present study.

This study demonstrated the efficacy of applying the BACI method of monitoring the impact of shoreline modification on near-shore communities, and the first known use of this approach to examine impacts of shoreline development on near-shore communities. The present study encompassed a thorough examination of community dynamics, including temporal and spatial data, and allowed inference of causal relationships from shoreline modification. The BACI design allows for causal mechanisms to be inferred as it follows treatments (in this case, impacted or control shorelines) that were subjected to the same background environmental conditions over time (before and after the shoreline modification), with shoreline modification as the difference between the treatments. In addition, BACI sampling can also be used to monitor the process of recolonization through changes in community composition in time, and it can allow for consideration of longer-term or larger-scale trends in site characteristics (*e.g.* salinity, DO, temperature, sediment grain size). The BACI design was enhanced through the use of linear model comparison, to examine changes using the most relevant models (of shoreline and other environmental factors), and determine outside drivers of particular influence to infaunal communities.

Several new research directions can be realized as a result of this study. Longer time-courses of study are necessary to examine the long-term responses of near-shore communities to shoreline modification and to determine the nature of the disturbance as pulse (short-term) impacts, or press (long-term, continued) impacts. A focus on additional environmental drivers and/or outside stressors within the context of a shoreline

development BACI study may help to elucidate community changes in response to these known drivers along with shoreline development. I suggest the management of shoreline erosion control strategies include monitoring of long-term effects, and if possible, avoid changes in sediment type to maintain community composition.

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**Table 1.** Numbers of replicates (n) of all BACI samples, infaunal suction and cores, crab scrapes, and seines at all four sites. Sample sizes of 500- $\mu$ m infauna differ from 3-mm infauna due to subsampling, number of samples sorted are in parentheses. Site abbreviations as follows: Tim = Timberneck, Dan = Dandy, Hol = Holly Cove, Win = Windy Hill. Dates under site names are the approximate date that shoreline modification project was completed in 2010. Dates = when samples were taken. Treat = shoreline treatments: C1 = control 1 shorelines, C2 = control 2, I1 = impact 1 (boat docks, Timberneck only), I2 = impact 2 (riprap, Timberneck only), I = impact.

Site	Year	Treat	Infauna: suction and cores			Crab Scrapes		Seines	
			n: 3 mm	n: 500 $\mu$ m	Dates	n	Dates	n	Dates
<b>TIM</b> 7/10	2010	C1	12	12 (7)	5/10, 5/12	12	6/2	5	6/8
		C2	12	12 (4)	5/11, 5/13	12	5/21, 6/2	6	6/10
		I1	3	12 (1)	5/10	2	6/2	2	6/7
		I2	3	12 (3)	5/10	4	6/2	3	6/7, 6/8
	2011	C1	12	12 (4)	5/6, 5/10	18	5/11, 5/18	6	5/11, 5/12
		C2	12	12 (6)	5/5	6	5/11	6	5/12
		I1	3	3 (2)	5/10	4	5/18	2	5/11
		I2	5	5 (2)	5/10	3	5/11, 5/18	5	5/11
<b>DAN</b> 6/10	2010	C1	6	6 (0)	5/20	4	5/21	2	6/11
		C2	11	11 (2)	5/19, 5/20	8	5/21	2	6/11
		I	6	6 (2)	5/19	6	5/21	0	-
	2011	C1	6	6 (0)	5/19	4	5/24	2	5/26
		C2	6	6 (2)	5/19	6	5/24	2	5/26
		I	6	6 (2)	5/19	8	5/24	2	5/26
<b>HOL</b> 6/25	2010	C1	4	4 (2)	6/14	4	6/14	3	6/21
		C2	6	6 (0)	6/14	6	6/14	5	6/21
		I	5	5 (2)	6/14	6	6/14	4	6/16
	2011	C1	5	5 (2)	6/6	5	6/6	2	6/7
		C2	3	3 (0)	6/6	4	6/6	2	6/7
		I	8	8 (2)	6/6	8	6/6	3	6/7
<b>WIN</b> 6/14	2010	C1	6	6 (2)	5/26	7	5/26	2	6/1
		C2	6	6 (0)	5/26	6	5/26	2	6/1
		I	6	6 (2)	5/26	6	5/26	2	6/1
	2011	C1	6	6 (2)	6/1	10	5/31	2	5/27
		C2	6	6 (0)	6/1	6	5/31	2	6/4
		I	6	6 (3)	6/1	6	5/31	3	5/27

**Table 2.** Candidate models and parameters for linear models to be compared with AICc analysis for the Timberneck Creek site with 2 impact groups. Each  $\beta$  symbol indicates the inclusion of that parameter in the model, and represents the coefficient associated with that parameter. K = the number of parameters for a given model (g1-g7), including  $\sigma^2$ .  $\beta_0$  indicates the reference condition, or intercept of the linear model, which is the mean of each response variable at control 1 shorelines in 2010. Year = 2011. Treat = shoreline treatment (control 2, impact 1, or impact 2). Y×T = year and treatment interaction. Sed = sediment grain size. Sal = salinity (PSU).

<b>Model</b>	<b>K</b>	<b>Intercept</b>	<b>Year</b>	<b>Treat</b>	<b>Y×T</b>	<b>Sed</b>	<b>Sal</b>
<b>g<sub>1</sub></b> (global)	11	$\beta_0$	$\beta_1$	$\beta_{2-4}$	$\beta_{5-7}$	$\beta_8$	$\beta_9$
<b>g<sub>2</sub></b> (Year+Treat)	6	$\beta_0$	$\beta_1$	$\beta_{2-4}$			
<b>g<sub>3</sub></b> (Year×Treat)	9	$\beta_0$	$\beta_1$	$\beta_{2-4}$	$\beta_{5-7}$		
<b>g<sub>4</sub></b> (Y×T+ Sed)	10	$\beta_0$	$\beta_1$	$\beta_{2-4}$	$\beta_{5-7}$	$\beta_8$	
<b>g<sub>5</sub></b> (Y×T + Sal)	10	$\beta_0$	$\beta_1$	$\beta_{2-4}$	$\beta_{5-7}$		$\beta_9$
<b>g<sub>6</sub></b> (Sed)	3	$\beta_0$				$\beta_8$	
<b>g<sub>7</sub></b> (Sal)	3	$\beta_0$					$\beta_9$



**Table 3.** Candidate models and parameters for linear models to be compared with AICc analysis for sites with 1 impact group: Dandy, Holly Cove, and Windy Hill. Each  $\beta$  symbol indicates the inclusion of that parameter in the model, and represents the coefficient associated with that parameter. K = the number of parameters for a given model (g1-g7), including  $\sigma^2$ .  $\beta_0$  indicates the reference condition, or intercept of the linear model, which is the mean of each response variable at control 1 shorelines in 2010. Year = 2011. Treat = shoreline treatment (control 2 or impact). Y×T= year and treatment interaction. Sed = sediment grain size. Sal = salinity (PSU).

Model	K	Intercept	Year	Treat	Y×T	Sed	Sal
<b>g<sub>1</sub></b> (global)	9	$\beta_0$	$\beta_1$	$\beta_{2-3}$	$\beta_{4-5}$	$\beta_6$	$\beta_7$
<b>g<sub>2</sub></b> (Year+Treat)	5	$\beta_0$	$\beta_1$	$\beta_{2-3}$			
<b>g<sub>3</sub></b> (Year×Treat)	7	$\beta_0$	$\beta_1$	$\beta_{2-3}$	$\beta_{4-5}$		
<b>g<sub>4</sub></b> (Y×T+ Sed)	8	$\beta_0$	$\beta_1$	$\beta_{2-3}$	$\beta_{4-5}$	$\beta_6$	
<b>g<sub>5</sub></b> (Y×T + Sal)	8	$\beta_0$	$\beta_1$	$\beta_{2-3}$	$\beta_{4-5}$		$\beta_7$
<b>g<sub>6</sub></b> (Sed)	3	$\beta_0$				$\beta_6$	
<b>g<sub>7</sub></b> (Sal)	3	$\beta_0$					$\beta_7$

**Table 4.** Mean values of water quality and sediment characteristics ( $\pm$  SE) by year and treatment. Treat = Treatment shorelines. C1 = control 1 shorelines. C2 = control 2 shorelines. I1 = impact 1 shorelines (boat docks, Timberneck only). I2 = impact 2 shorelines (riprap, Timberneck only). I = impact shorelines. % Coarse = percent of coarse sediments: sand + gravel. Site abbreviations as in Table 1.

Site	Year	Treat	n	% Coarse	Salinity (PSU)	DO (mg/L)	Temp (°C)
TIM	2010	C1	12	45.78 (9.73)	14.93 (0.09)	8.28 (0.35)	21.37 (0.49)
		C2	12	59.31 (5.77)	15.30 (0.20)	8.88 (0.39)	18.55 (0.40)
		I1	3	92.65 (0.86)	14.49 (0.10)	7.30 (7e-08)	20.97 (0.30)
		I2	3	46.70 (13.31)	14.99 (0.31)	7.93 (0.23)	20.20 (0.35)
	2011	C1	12	61.15 (8.53)	14.33 (0.18)	9.59 (0.46)	20.19 (0.44)
		C2	12	59.59 (8.39)	13.76 (0.07)	9.68 (0.19)	18.38 (0.24)
		I1	3	91.69 (2.70)	14.93 (0.09)	8.07 (0.20)	21.03 (0.20)
		I2	5	91.25 (2.67)	14.94 (0.11)	8.16 (0.09)	21.38 (0.13)
DAN	2010	C1	6	97.53 (0.33)	21.33 (0.16)	9.00 (0.19)	16.98 (0.06)
		C2	11	94.40 (0.80)	19.69 (0.44)	8.94 (0.37)	15.98 (0.62)
		I	6	96.27 (0.34)	19.5 (0.09)	9.93 (0.22)	16.63 (0.01)
	2011	C1	6	94.42 (2.18)	15.36 (0.02)	8.70 (0.15)	20.98 (0.05)
		C2	6	96.85 (0.60)	15.19 (0.004)	8.14 (0.17)	21.2 (0.34)
		I	6	95.67 (0.42)	15.16 (1e-07)	7.75 (0.02)	20.9 (0)
HOL	2010	C1	4	88.85 (0.65)	14.17 (0.02)	9.55 (0.55)	31.38 (0.10)
		C2	6	82.68 (8.12)	14.19 (0.01)	8.63 (0.21)	30.60 (0.10)
		I	5	92.36 (4.08)	14.16 (0.01)	9.10 (0.23)	31.36 (0.09)
	2011	C1	5	82.57 (1.91)	9.79 (0.05)	7.29 (0.31)	28.98 (0.07)
		C2	3	79.73 (6.15)	9.48 (0.34)	8.20 (0.25)	28.93 (0.09)
		I	8	97.16 (0.29)	9.74 (0.06)	7.66 (0.25)	28.18 (0.14)
WIN	2010	C1	6	98.31 (0.35)	5.38 (0.04)	15.45 (0.64)	25.92 (0.60)
		C2	6	95.85 (1.69)	5.4 (0.05)	14.08 (0.31)	26.97 (0.28)
		I	6	98.47 (0.22)	5.38 (0.02)	13.52 (0.24)	26.52 (0.15)
	2011	C1	6	96.49 (0.88)	3.5 (0.06)	7.43 (0.50)	30.60 (0.32)
		C2	6	84.84 (9.12)	3.53 (0.13)	7.00 (1.12)	30.17 (0.59)
		I	6	84.67 (4.46)	3.62 (0.04)	6.47 (0.22)	30.32 (0.20)

**Table 5.** AIC results for all models of 3-mm infauna at Timberneck Creek, ordered by decreasing Akaike weight ( $w_i$ ). Indications of data transformations are in parentheses under response variable name. Bolded rows indicate models with strong support ( $w_i \geq 0.1$ ). Abbreviations for models and K as in table 2.  $R^2$  and Adjusted  $R^2$  (Adj.  $R^2$ ) are also listed for each model.

Timberneck 3-mm							
Response	Model	K	AICc	$\Delta$ AICc	Weight ( $w_i$ )	$R^2$	Adj. $R^2$
Density (Box-Cox)	<b>g<sub>2</sub></b>	<b>6</b>	<b>263.8</b>	<b>0</b>	<b>0.875</b>	<b>0.204</b>	<b>0.148</b>
	g <sub>3</sub>	9	270.1	6.23	0.039	0.225	0.125
	g <sub>6</sub>	3	270.1	6.31	0.037	0.011	-0.005
	g <sub>7</sub>	3	270.8	7.01	0.026	1.3E-04	-0.017
	g <sub>5</sub>	10	272.7	8.84	0.011	0.229	0.112
	g <sub>4</sub>	10	272.8	8.93	0.01	0.227	0.111
	g <sub>1</sub>	11	275.5	11.67	0.003	0.230	0.097
Biomass (Box-Cox)	<b>g<sub>1</sub></b>	<b>11</b>	<b>149.9</b>	<b>0</b>	<b>0.619</b>	<b>0.546</b>	<b>0.468</b>
	<b>g<sub>4</sub></b>	<b>10</b>	<b>151.2</b>	<b>1.27</b>	<b>0.327</b>	<b>0.514</b>	<b>0.441</b>
	g <sub>6</sub>	3	154.8	4.87	0.054	0.313	0.301
	g <sub>2</sub>	6	169.7	19.85	0	0.220	0.165
	g <sub>3</sub>	9	173.4	23.47	0	0.272	0.178
	g <sub>5</sub>	10	174.3	24.43	0	0.294	0.188
	g <sub>7</sub>	3	176.5	26.65	0	0.023	0.007
Diversity (logit)	<b>g<sub>6</sub></b>	<b>3</b>	<b>188.4</b>	<b>0</b>	<b>0.485</b>	<b>0.305</b>	<b>0.293</b>
	<b>g<sub>4</sub></b>	<b>10</b>	<b>188.7</b>	<b>0.34</b>	<b>0.409</b>	<b>0.476</b>	<b>0.397</b>
	<b>g<sub>1</sub></b>	<b>11</b>	<b>191.4</b>	<b>3.07</b>	<b>0.105</b>	<b>0.673</b>	<b>0.616</b>
	g <sub>2</sub>	6	201.3	12.96	0.001	0.236	0.182
	g <sub>3</sub>	9	205.9	17.56	0	0.276	0.182
	g <sub>7</sub>	3	206.4	18.08	0	0.281	0.173
	g <sub>5</sub>	10	208.3	19.96	0	0.069	0.039

**Table 6.** Parameter estimates for Timberneck Creek 3-mm infauna from the models supported with  $w_i \geq 0.1$ . Parameter estimates with 95% confidence intervals that do not include 0 (estimates significant at  $\alpha = 0.05$ ) are in bold, and with 80% confidence intervals (for interaction estimates only) that do not include 0 ( $\alpha = 0.2$ ) in italics. Xs are parameters not included in the selected models. Abbreviations for models as in table 2, candidate models. Year = 2011. Treat = shoreline treatment (C2 = control 2, I1 = impact 1, or I2 = impact 2). Y×T = year and treatment interaction for the shoreline treatment in parentheses. Sed = sediment grain size as % coarse (sand + gravel) sediments. Sal = salinity (PSU).

Timberneck 3-mm		Parameter Estimates ( $\pm$ SE)									
Response	Model	$\beta_0$	$\beta_1$	$\beta_2$	$\beta_3$	$\beta_4$	$\beta_5$	$\beta_6$	$\beta_7$	$\beta_8$	$\beta_9$
		Intercept	Year (2011)	Treat C2	Treat I1	Treat I2	Y×T (C2)	Y×T (I1)	Y×T (I2)	Sed	Sal
Density (Box-Cox)	g2	<b>5.78</b> ± <b>0.46</b>	0.21 ± 0.48	<b>1.61</b> ± <b>0.55</b>	<b>2.76</b> ± <b>0.87</b>	0.69 ± 0.78	X	X	X	X	X
Biomass (Box-Cox)	g1	-4.99 ± 3.02	0.24 ± 0.32	0.21 ± 0.30	0.38 ± 0.50	<b>-0.93</b> ± <b>0.46</b>	-0.06 ± 0.45	0.55 ± 0.69	-0.14 ± 0.61	<b>-0.02</b> ± <b>0.004</b>	0.39 ± 0.20
	g4	<b>0.79</b> ± <b>0.27</b>	-2e-04 ± 0.30	0.35 ± 0.30	0.19 ± 0.50	-0.91 ±0.47	-0.42 ± 0.42	0.96 ± 0.67	0.06 ± 0.62	<b>-0.02</b> ± <b>0.004</b>	X
Diversity (logit)	g6	<b>-1.45</b> ± <b>0.32</b>	X	X	X	X	X	X	X	<b>0.02</b> ± <b>0.005</b>	X
	g4	<b>-1.83</b> ± <b>0.37</b>	<b>0.88</b> ± <b>0.41</b>	-0.07 ± 0.41	0.86 ± 0.68	0.12 ± 0.64	0.41 ± 0.57	-1.30 ±0.90	-1.01 ± 0.84	<b>0.02</b> ± <b>0.005</b>	X
	g1	-3.70 ± 4.22	<b>0.96</b> ± <b>0.45</b>	-0.11 ± 0.42	0.92 ± 0.70	0.11 ± 0.64	0.52 ± 0.63	-1.43 ±0.96	-1.08 ± 0.86	<b>0.02</b> ± <b>0.005</b>	0.13 ± 0.28

**Table 7.** AIC results for all models of 500- $\mu\text{m}$  infauna at Timberneck Creek, ordered by decreasing Akaike weight ( $w_i$ ). Indications of data transformations are in parentheses under response variable name. Bolded rows indicate models with strong support ( $w_i \geq 0.1$ ). Abbreviations for models and K as in table 2.  $R^2$  and Adjusted  $R^2$  (Adj.  $R^2$ ) are also listed for each model.

Timberneck 500- $\mu\text{m}$ infauna							
Response	Model	K	AICc	$\Delta\text{AICc}$	Weight ( $w_i$ )	$R^2$	Adj. $R^2$
Density (raw)	<b>g<sub>3</sub></b>	<b>7</b>	<b>306.8</b>	<b>0</b>	<b>0.381</b>	<b>0.649</b>	<b>0.573</b>
	<b>g<sub>4</sub></b>	<b>8</b>	<b>307.3</b>	<b>0.47</b>	<b>0.302</b>	<b>0.688</b>	<b>0.603</b>
	<b>g<sub>5</sub></b>	<b>8</b>	<b>307.8</b>	<b>1.03</b>	<b>0.227</b>	<b>0.682</b>	<b>0.595</b>
	g <sub>1</sub>	9	310.1	3.3	0.073	0.703	0.640
	g <sub>2</sub>	5	313.1	6.29	0.016	0.451	0.385
	g <sub>6</sub>	3	320	13.17	0.001	0.153	0.122
	g <sub>7</sub>	3	320.8	14.02	0.000	0.128	0.096
Biomass (Box-Cox)	<b>g<sub>7</sub></b>	<b>3</b>	<b>42.5</b>	<b>0</b>	<b>0.854</b>	<b>0.206</b>	<b>0.176</b>
	g <sub>3</sub>	7	48.1	5.63	0.051	0.371	0.234
	g <sub>2</sub>	5	48.3	5.86	0.046	0.200	0.104
	g <sub>6</sub>	3	49	6.49	0.033	0.007	-0.030
	g <sub>4</sub>	8	51.9	9.41	0.008	0.372	0.201
	g <sub>5</sub>	8	51.9	9.45	0.008	0.372	0.200
	g <sub>1</sub>	9	56	13.57	0.001	0.375	0.141
Diversity (logit)	<b>g<sub>6</sub></b>	<b>3</b>	<b>85.8</b>	<b>0</b>	<b>0.738</b>	<b>0.270</b>	<b>0.243</b>
	g <sub>2</sub>	5	90.2	4.43	0.081	0.300	0.216
	g <sub>4</sub>	8	91.1	5.31	0.052	0.499	0.363
	g <sub>5</sub>	8	91.2	5.39	0.050	0.498	0.361
	g <sub>3</sub>	7	92	6.17	0.034	0.411	0.283
	g <sub>1</sub>	9	92.7	6.92	0.023	0.543	0.391
	g <sub>7</sub>	3	92.7	6.94	0.023	0.073	0.039

**Table 8.** Parameter estimates for Timberneck Creek 500- $\mu\text{m}$  infauna from the models supported with  $w_i \geq 0.1$ . Parameter estimates with 95% confidence intervals that do not include 0 (estimates significant at  $\alpha = 0.05$ ) are in bold, and with 80% confidence intervals (for interaction estimates only) that do not include 0 ( $\alpha = 0.2$ ) in italics. Xs are parameters not included in the selected models. Abbreviations for models as in table 2, candidate models. Year = 2011. Treat = shoreline treatment (C2 = control 2, I = impact). Y $\times$ T = year and treatment interaction for the shoreline treatment in parentheses. Sed = sediment grain size as % coarse (sand + gravel) sediments. Sal = salinity (PSU).

Timberneck 500- $\mu\text{m}$ infauna		Parameter Estimates ( $\pm$ SE)							
Response	Model	$\beta_0$ Intercept	$\beta_1$ Year (2011)	$\beta_2$ Treat C2	$\beta_3$ Treat I	$\beta_4$ Y $\times$ T (C2)	$\beta_5$ Y $\times$ T (I)	$\beta_6$ Sed	$\beta_7$ Sal
Density (raw)	$g_3$	<b>147.14</b> $\pm$ <b>14.59</b>	<b>-129.89</b> <b><math>\pm</math> 24.19</b>	<b>-96.14</b> $\pm$ <b>24.19</b>	-44.14 $\pm$ 24.19	<b>124.39</b> $\pm$ <b>34.73</b>	<i>74.64</i> $\pm$ <i>36.48</i>	X	X
	$g_4$	<b>164.77</b> $\pm$ <b>17.65</b>	<b>-125.91</b> <b><math>\pm</math> 23.45</b>	<b>-88.21</b> $\pm$ <b>23.82</b>	-34.32 $\pm$ 24.07	<b>123.51</b> $\pm$ <b>33.49</b>	<b>85.63</b> $\pm$ <b>35.79</b>	-0.47 $\pm$ 0.29	X
	$g_5$	-195.25 $\pm$ 288.19	<b>-116.3</b> <b><math>\pm</math> 25.23</b>	<b>-109.33</b> <b><math>\pm</math> 25.14</b>	-40.06 $\pm$ 23.71	<b>150.31</b> $\pm$ <b>37.96</b>	<i>61.45</i> $\pm$ <i>36.58</i>	X	22.79 $\pm$ 15.16
Biomass (Box-Cox)	$g_7$	<b>-7.41</b> $\pm$ <b>1.78</b>	X	X	X	X	X	X	<b>0.32</b> $\pm$ <b>0.12</b>
Diversity (logit)	$g_6$	<b>-1.37</b> $\pm$ <b>0.39</b>	X	X	X	X	X	<b>0.02</b> $\pm$ <b>0.006</b>	X

**Table 9.** AIC results for all models of 3-mm infauna at Dandy, ordered by decreasing Akaike weight ( $w_i$ ). Indications of data transformations are in parentheses under response variable name. Bolded rows indicate models with strong support ( $w_i \geq 0.1$ ). Abbreviations for models and K as in table 3.  $R^2$  and Adjusted  $R^2$  (Adj.  $R^2$ ) are also listed for each model.

Dandy 3-mm							
Response	Model	K	AICc	$\Delta$ AICc	Weight ( $w_i$ )	$R^2$	Adj. $R^2$
Density (Box-Cox)	<b>g<sub>6</sub></b>	<b>3</b>	<b>29.0</b>	<b>0</b>	<b>0.564</b>	<b>0.033</b>	<b>0.008</b>
	<b>g<sub>7</sub></b>	<b>3</b>	<b>30.4</b>	<b>1.36</b>	<b>0.286</b>	<b>0.001</b>	<b>-0.025</b>
	<b>g<sub>2</sub></b>	<b>5</b>	<b>31.9</b>	<b>2.85</b>	<b>0.135</b>	<b>0.084</b>	<b>0.010</b>
	g <sub>3</sub>	7	37.1	8.12	0.01	0.093	-0.036
	g <sub>4</sub>	8	38.9	9.88	0.004	0.123	-0.032
Biomass (Box-Cox)	<b>g<sub>6</sub></b>	<b>3</b>	<b>145.3</b>	<b>0</b>	<b>0.56</b>	<b>0.017</b>	<b>-0.008</b>
	<b>g<sub>7</sub></b>	<b>3</b>	<b>146.0</b>	<b>0.7</b>	<b>0.394</b>	<b>0.000</b>	<b>-0.026</b>
	g <sub>2</sub>	5	150.5	5.2	0.042	0.014	-0.066
	g <sub>3</sub>	7	155.5	10.21	0.003	0.030	-0.109
	g <sub>4</sub>	8	158.2	12.88	0.001	0.400	-0.130
Diversity (logit)	<b>g<sub>2</sub></b>	<b>5</b>	<b>113.9</b>	<b>0</b>	<b>0.495</b>	<b>0.173</b>	<b>0.106</b>
	<b>g<sub>7</sub></b>	<b>3</b>	<b>115.3</b>	<b>1.36</b>	<b>0.251</b>	<b>0.032</b>	<b>0.007</b>
	<b>g<sub>6</sub></b>	<b>3</b>	<b>115.8</b>	<b>1.82</b>	<b>0.2</b>	<b>0.021</b>	<b>-0.004</b>
	g <sub>3</sub>	7	119.1	5.11	0.038	0.184	0.067
	g <sub>2</sub>	8	120.8	6.86	0.016	0.211	0.071

**Table 10.** Parameter estimates for Dandy 3-mm infauna from the models supported with  $w_i \geq 0.1$ . Parameter estimates with 95% confidence intervals that do not include 0 (estimates significant at  $\alpha = 0.05$ ) are in bold, and with 80% confidence intervals (for interaction estimates only) that do not include 0 ( $\alpha = 0.2$ ) in italics. Xs are parameters not included in the selected models. Abbreviations for models as in table 3, candidate models. Year = 2011. Treat = shoreline treatment (C2 = control 2, I = impact). Y×T = year and treatment interaction for the shoreline treatment in parentheses. Sed = sediment grain size as % coarse (sand + gravel) sediments. Sal = salinity (PSU).

Dandy 3-mm		Parameter Estimates ( $\pm$ SE)							
Response	Model	$\beta_0$ Intercept	$\beta_1$ Year (2011)	$\beta_2$ Treat C2	$\beta_3$ Treat I	$\beta_4$ Y×T (C2)	$\beta_5$ Y×T (I)	$\beta_6$ Sed	$\beta_7$ Sal
Density (Box-Cox)	g <sub>6</sub>	<b>4.40 ± 1.82</b>	X	X	X	X	X	-0.02 ± 0.02	X
	g <sub>7</sub>	<b>2.48 ± 1.10</b>	X	X	X	X	X	X	-0.01 ± 0.07
	g <sub>2</sub>	<b>2.45 ± 0.11</b>	-0.01 ± 0.10	-0.19 ± 0.13	-0.24 ± 0.14	X	X	X	X
Biomass (Box-Cox)	g <sub>6</sub>	6.21 ± 7.51	X	X	X	X	X	-0.06 ± 0.08	X
	g <sub>7</sub>	-0.17 ± 4.49	X	X	X	X	X	X	0.01 ± 0.28
Diversity (logit)	g <sub>2</sub>	<b>1.15 ± 0.31</b>	0.29 ± 0.31	-0.54 ± 0.34	0.21 ± 0.37	X	X	X	X
	g <sub>7</sub>	4.61 ± 3.09	X	X	X	X	X	X	-0.22 ± 0.19
	g <sub>6</sub>	5.92 ± 5.24	X	X	X	X	X	-0.05 ± 0.05	X



**Table 11.** Parameter estimates for Dandy 500- $\mu\text{m}$  infauna from the linear model (similar to  $g_3$  for 3-mm infauna). Parameter estimates with 95% confidence intervals that do not include 0 (estimates significant at  $\alpha = 0.05$ ) are in bold, and with 80% confidence intervals (for interaction estimates only) that do not include 0 ( $\alpha = 0.2$ ) in italics. Xs are parameters not included in the model. Year = 2011. Treat = shoreline treatment (I = impact). Y $\times$ T = year and treatment interaction for the shoreline treatment in parentheses.

Dandy 500- $\mu\text{m}$ infauna		Parameter Estimates ( $\pm$ SE)			
Response	Adj. R <sup>2</sup>	$\beta_0$ Intercept	$\beta_1$ Year (2011)	$\beta_2$ Treat I	$\beta_3$ Y $\times$ T (I)
Density (raw)	0.238	<b>78 <math>\pm</math> 13.87</b>	-34.5 $\pm$ 19.62	-6.5 $\pm$ 19.62	6.5 $\pm$ 27.74
Biomass (Box-Cox)	0.262	<b>-3.91 <math>\pm</math> 0.93</b>	-3.04 $\pm$ 1.31	-1.84 $\pm$ 1.31	<i>3.45 <math>\pm</math> 1.86</i>
Diversity (logit)	0.226	0.63 $\pm$ 0.43	-1.16 $\pm$ 0.61	-0.20 $\pm$ 0.61	0.46 $\pm$ 0.86

**Table 12.** AIC results for all models of 3-mm infauna at Holly Cove, ordered by decreasing Akaike weight ( $w_i$ ). Indications of data transformations are in parentheses under response variable name. Bolded rows indicate models with strong support ( $w_i \geq 0.1$ ). Abbreviations for models and K as in table 2.  $R^2$  and Adjusted  $R^2$  (Adj.  $R^2$ ) are also listed for each model.

Holly Cove 3-mm							
Response	Model	K	AICc	$\Delta$ AICc	Weight ( $w_i$ )	$R^2$	Adj. $R^2$
Density (Box-Cox)	<b>g<sub>4</sub></b>	<b>8</b>	<b>137.3</b>	<b>0</b>	<b>0.573</b>	<b>0.599</b>	<b>0.498</b>
	<b>g<sub>6</sub></b>	<b>3</b>	<b>138.0</b>	<b>0.68</b>	<b>0.409</b>	<b>0.320</b>	<b>0.297</b>
	g <sub>7</sub>	3	145.0	7.63	0.013	0.150	0.120
	g <sub>2</sub>	5	146.9	9.53	0.005	0.243	0.159
	g <sub>3</sub>	7	150.2	12.87	0.001	0.316	0.179
Biomass (Box-Cox)	<b>g<sub>6</sub></b>	<b>3</b>	<b>120.0</b>	<b>0</b>	<b>0.893</b>	<b>0.396</b>	<b>0.376</b>
	<b>g<sub>4</sub></b>	<b>8</b>	<b>124.3</b>	<b>4.26</b>	<b>0.106</b>	<b>0.582</b>	<b>0.478</b>
	g <sub>7</sub>	3	134.3	14.28	0.001	0.043	0.010
	g <sub>2</sub>	5	137.9	17.85	0	0.101	0.001
	g <sub>3</sub>	7	140.7	20.7	0	0.200	0.040
Diversity (logit)	<b>g<sub>6</sub></b>	<b>3</b>	<b>71.4</b>	<b>0</b>	<b>0.689</b>	<b>0.122</b>	<b>0.091</b>
	<b>g<sub>2</sub></b>	<b>5</b>	<b>73.8</b>	<b>2.44</b>	<b>0.203</b>	<b>0.204</b>	<b>0.116</b>
	g <sub>7</sub>	3	75.4	4.02	0.092	2.0E-05	-0.034
	g <sub>3</sub>	7	79.6	8.16	0.012	0.223	0.062
	g <sub>4</sub>	8	81.5	10.06	0.005	0.267	0.083

**Table 13.** Parameter estimates for Holly Cove 3-mm infauna from the models supported with  $w_i \geq 0.1$ . Parameter estimates with 95% confidence intervals that do not include 0 (estimates significant at  $\alpha = 0.05$ ) are in bold, and with 80% confidence intervals (for interaction estimates only) that do not include 0 ( $\alpha = 0.2$ ) in italics. Xs are parameters not included in the selected models. Abbreviations for models as in table 3, candidate models. Year = 2011. Treat = shoreline treatment (C2 = control 2, I = impact). Y×T = year and treatment interaction for the shoreline treatment in parentheses. Sed = sediment grain size as % coarse (sand + gravel) sediments. Sal = salinity (PSU).

Holly Cove 3-mm		Parameter Estimates ( $\pm$ SE)							
Response	Model	$\beta_0$ Intercept	$\beta_1$ Year (2011)	$\beta_2$ Treat C2	$\beta_3$ Treat I	$\beta_4$ Y×T (C2)	$\beta_5$ Y×T (I)	$\beta_6$ Sed	$\beta_7$ Sal
Density (Box-Cox)	g <sub>4</sub>	-5.07 ± 3.16	<b>-3.09 ±</b> <b>1.19</b>	-0.04 ± 1.15	-0.08 ± 1.18	2.96 ± <i>1.71</i>	0.79 ± 1.56	<b>0.14 ±</b> <b>0.03</b>	X
	g <sub>6</sub>	-4.62 ± 2.96	X	X	X	X	X	<b>0.12 ±</b> <b>0.03</b>	X
Biomass (Box-Cox)	g <sub>6</sub>	<b>-12.98 ±</b> <b>2.21</b>	X	X	X	X	X	<b>0.11 ±</b> <b>0.02</b>	X
	g <sub>4</sub>	<b>-15.26 ±</b> <b>2.56</b>	0.32 $\pm 0.97$	<b>2.08 ±</b> <b>0.93</b>	0.25 ± 0.95	-2.83 ± <i>1.39</i>	-0.83 ± 1.29	<b>0.13 ±</b> <b>0.03</b>	X
Diversity (logit)	g <sub>6</sub>	-1.93 ± 1.01	X	X	X	X	X	0.02 ± 0.01	X
	g <sub>2</sub>	-0.39 ± 0.27	-0.04 ± 0.26	0.53 ± 0.33	<b>0.795</b> $\pm 0.30$	X	X	X	X

**Table 14.** Parameter estimates for Holly Cove 500- $\mu\text{m}$  infauna from the linear model (similar to  $g_3$  for 3-mm infauna). Parameter estimates with 95% confidence intervals that do not include 0 (estimates significant at  $\alpha = 0.05$ ) are in bold, and with 80% confidence intervals (for interaction estimates only) that do not include 0 ( $\alpha = 0.2$ ) in italics. Xs are parameters not included in the model. Year = 2011. Treat = shoreline treatment (I = impact). Y $\times$ T = year and treatment interaction for the shoreline treatment in parentheses.

Holly Cove 500- $\mu\text{m}$ infauna		Parameter Estimates ( $\pm$ SE)			
Response	Adj. R <sup>2</sup>	$\beta_0$ Intercept	$\beta_1$ Year (2011)	$\beta_2$ Treat I	$\beta_3$ Y $\times$ T (I)
Density (raw)	0.703	11.5 $\pm$ 5.41	-3.0 $\pm$ 7.65	2 $\pm$ 7.65	28 $\pm$ 10.82
Biomass (Box-Cox)	-0.358	<b>-2.54 <math>\pm</math> 0.13</b>	0.01 $\pm$ 0.18	-0.11 $\pm$ 0.18	0.18 $\pm$ 0.25
Diversity (logit)	-0.162	0.95 $\pm$ 0.55	0.12 $\pm$ 0.77	-0.87 $\pm$ 0.77	0.66 $\pm$ 1.10

**Table 15.** AIC results for all models of 3-mm infauna at Windy Hill, ordered by decreasing Akaike weight ( $w_i$ ). Indications of data transformations are in parentheses under response variable name. Bolded rows indicate models with strong support ( $w_i \geq 0.1$ ). Abbreviations for models and K as in table 2.  $R^2$  and Adjusted  $R^2$  (Adj.  $R^2$ ) are also listed for each model.

Windy Hill 3-mm							
Response	Model	K	AICc	$\Delta$ AICc	Weight ( $w_i$ )	$R^2$	Adj. $R^2$
Density (raw)	<b>g<sub>7</sub></b>	<b>3</b>	<b>332.4</b>	<b>0</b>	<b>0.669</b>	<b>0.171</b>	<b>0.146</b>
	<b>g<sub>2</sub></b>	<b>5</b>	<b>334.5</b>	<b>2.1</b>	<b>0.234</b>	<b>0.240</b>	<b>0.169</b>
	g <sub>3</sub>	7	337.6	5.22	0.049	0.298	0.181
	g <sub>4</sub>	8	339.0	6.62	0.024	0.335	0.198
	g <sub>6</sub>	3	339.1	6.71	0.023	0.000	-0.029
Biomass (raw)	<b>g<sub>4</sub></b>	<b>8</b>	<b>45.2</b>	<b>0</b>	<b>0.344</b>	<b>0.486</b>	<b>0.380</b>
	<b>g<sub>7</sub></b>	<b>3</b>	<b>45.3</b>	<b>0.13</b>	<b>0.323</b>	<b>0.227</b>	<b>0.204</b>
	<b>g<sub>3</sub></b>	<b>7</b>	<b>46.0</b>	<b>0.77</b>	<b>0.235</b>	<b>0.424</b>	<b>0.328</b>
	g <sub>2</sub>	5	47.8	2.58	0.095	0.284	0.217
	g <sub>6</sub>	3	54.6	9.38	0.003	2.0E-05	-0.029
Diversity (logit)	<b>g<sub>3</sub></b>	<b>7</b>	<b>92.3</b>	<b>0</b>	<b>0.485</b>	<b>0.394</b>	<b>0.293</b>
	<b>g<sub>2</sub></b>	<b>5</b>	<b>93.6</b>	<b>1.35</b>	<b>0.248</b>	<b>0.256</b>	<b>0.187</b>
	<b>g<sub>4</sub></b>	<b>8</b>	<b>94.2</b>	<b>1.98</b>	<b>0.18</b>	<b>0.416</b>	<b>0.295</b>
	g <sub>7</sub>	3	96.6	4.33	0.056	0.065	0.038
	g <sub>6</sub>	3	97.8	5.52	0.031	0.034	0.005

**Table 16.** Parameter estimates for Windy Hill 3-mm infauna from the models supported with  $w_i \geq 0.1$ . Parameter estimates with 95% confidence intervals that do not include 0 (estimates significant at  $\alpha = 0.05$ ) are in bold, and with 80% confidence intervals (for interaction estimates only) that do not include 0 ( $\alpha = 0.2$ ) in italics. Xs are parameters not included in the selected models. Abbreviations for models as in table 3, candidate models. Year = 2011. Treat = shoreline treatment (C2 = control 2, I = impact). Y×T = year and treatment interaction for the shoreline treatment in parentheses. Sed = sediment grain size as % coarse (sand + gravel) sediments. Sal = salinity (PSU).

Windy Hill 3-mm		Parameter Estimates ( $\pm$ SE)							
Response	Model	$\beta_0$ Intercept	$\beta_1$ Year (2011)	$\beta_2$ Treat C2	$\beta_3$ Treat I	$\beta_4$ Y×T (C2)	$\beta_5$ Y×T (I)	$\beta_6$ Sed	$\beta_7$ Sal
Density (raw)	g7	-18.05 ± 18.74	X	X	X	X	X	X	<b>10.85</b> ± 4.11
	g2	<b>31.61</b> ± <b>7.54</b>	-19.89 ± 7.54	16.25 ± 9.24	10.08 ± 9.24	X	X	X	X
Biomass (raw)	g4	-0.13 ± 0.67	0.02 ± 0.22	<b>0.57</b> ± <b>0.22</b>	<b>0.48</b> ± <b>0.22</b>	<b>-0.75</b> ± <b>0.31</b>	<b>-0.97</b> ± <b>0.32</b>	-0.01 ± 0.007	X
	g7	<b>-2.28</b> ± <b>0.35</b>	X	X	X	X	X	X	0.24 ± 0.08
	g3	<b>-1.35</b> ± <b>0.16</b>	0.05 ± 0.23	<b>0.60</b> ± <b>0.23</b>	<b>0.48</b> ± <b>0.23</b>	<i>-0.63</i> ± <i>0.32</i>	<b>-0.82</b> ± <b>0.32</b>	X	X
Diversity (logit)	g3	-0.39 ± 0.30	0.87 ± 0.43	0.37 ± 0.43	0.14 ± 0.43	0.06 ± 0.61	<b>-1.34</b> ± <b>0.61</b>	X	X
	g2	-0.18 ± 0.27	0.45 ± 0.27	0.40 ± 0.33	-0.53 ± 0.33	X	X	X	X
	g4	0.96 ± 1.32	0.85 ± 0.43	0.34 ± 0.43	0.14 ± 0.43	-0.06 ± 0.62	<b>-1.50</b> ± <b>0.63</b>	-0.01 ± 0.01	X

**Table 17.** Parameter estimates for Windy Hill 500- $\mu\text{m}$  infauna from the linear model (similar to  $g_3$  for 3-mm infauna). Parameter estimates with 95% confidence intervals that do not include 0 (estimates significant at  $\alpha = 0.05$ ) are in bold, and with 80% confidence intervals (for interaction estimates only) that do not include 0 ( $\alpha = 0.2$ ) in italics. Xs are parameters not included in the model. Year = 2011. Treat = shoreline treatment (I = impact). Y $\times$ T = year and treatment interaction for the shoreline treatment in parentheses.

Windy Hill 500- $\mu\text{m}$ infauna		Parameter Estimates ( $\pm$ SE)			
Response	Adj. $R^2$	$\beta_0$ Intercept	$\beta_1$ Year (2011)	$\beta_2$ Treat I	$\beta_3$ Y $\times$ T (I)
Density (Box-Cox)	-0.213	<b>5.90 <math>\pm</math> 1.24</b>	1.57 $\pm$ 1.76	1.48 $\pm$ 1.76	-2.99 $\pm$ 2.38
Biomass (raw)	0.186	<b>0.02 <math>\pm</math> 0.009</b>	0.0002 $\pm$ 0.01	0.015 $\pm$ 0.01	<i>-0.025 <math>\pm</math> 0.016</i>
Diversity (logit)	0.369	-0.03 $\pm$ 0.31	-0.14 $\pm$ 0.44	0.92 $\pm$ 0.44	-0.26 $\pm$ 0.59

**Table 18.** Parameter estimates for Timberneck predators from the linear model (similar to  $g_3$  for 3-mm infauna). Parameter estimates with 95% confidence intervals that do not include 0 (estimates significant at  $\alpha = 0.05$ ) are in bold, and with 80% confidence intervals (for interaction estimates only) that do not include 0 ( $\alpha = 0.2$ ) in italics. Xs are parameters not included in the model. Year = 2011. Treat = shoreline treatment (C2 = control 2, I = impact). Y×T = year and treatment interaction for the shoreline treatment in parentheses.

Timberneck Predators		Parameter Estimates ( $\pm$ SE)							
Response	Adj. $R^2$	$\beta_0$ Intercept	$\beta_1$ Year (2011)	$\beta_2$ Treat C2	$\beta_3$ Treat I1	$\beta_4$ Treat I2	$\beta_5$ Y×T (C2)	$\beta_6$ Y×T (I1)	$\beta_7$ Y×T (I2)
Blue Crab Abundance (Box-Cox)	0.054	<b>1.37</b> ± <b>0.49</b>	0.22 ± 0.63	0.83 ± 0.69	-1.99 ± 1.28	-1.27 ± 0.97	-0.34 ± 1.05	2.29 ± 1.58	2.43 ± 1.43
Fish Abundance (Box-Cox)	0.394	<b>7.26</b> ± <b>0.55</b>	<b>-1.95</b> ± <b>0.74</b>	0.10 ± 0.74	-1.08 ± 1.03	-0.30 ± 0.90	-0.17 ± 1.03	1.25 ± 1.44	-0.77 ± 1.17
Fish Diversity (logit)	0.057	-0.17 ± 0.33	-0.57 ± 0.45	-0.21 ± 0.45	0.13 ± 0.62	0.10 ± 0.54	-6E-05 ± 0.62	-0.31 ± 0.86	0.70 ± 0.70



**Table 19.** Parameter estimates for Dandy predators from the linear model (similar to  $g_3$  for 3-mm infauna). Parameter estimates with 95% confidence intervals that do not include 0 (estimates significant at  $\alpha = 0.05$ ) are in bold, and with 80% confidence intervals (for interaction estimates only) that do not include 0 ( $\alpha = 0.2$ ) in italics. Xs are parameters not included in the model. Year = 2011. Treat = shoreline treatment (C2 = control 2, I = impact). Y×T = year and treatment interaction for the shoreline treatment in parentheses. Fish abundance and diversity at Dandy were not modeled due to a lack of seines in 2010 at impact sites.

Dandy Predators		Parameter Estimates ( $\pm$ SE)					
Response	Adj. R <sup>2</sup>	$\beta_0$	$\beta_1$	$\beta_2$	$\beta_3$	$\beta_4$	$\beta_5$
		Intercept	Year (2011)	Treat C2	Treat I	Y×T (C2)	Y×T (I)
Blue Crab Abundance (Box-Cox)	0.084	0.33 ± 1.08	-2.14 ± 1.53	1.30 ± 1.33	1.38 ± 1.40	1.37 ± 1.93	1.16 ± 1.93

**Table 20.** Parameter estimates for Holly Cove predators from the linear model (similar to  $g_3$  for 3-mm infauna). Parameter estimates with 95% confidence intervals that do not include 0 (estimates significant at  $\alpha = 0.05$ ) are in bold, and with 80% confidence intervals (for interaction estimates only) that do not include 0 ( $\alpha = 0.2$ ) in italics. Xs are parameters not included in the model. Year = 2011. Treat = shoreline treatment (C2 = control 2, I = impact). Y×T = year and treatment interaction for the shoreline treatment in parentheses.

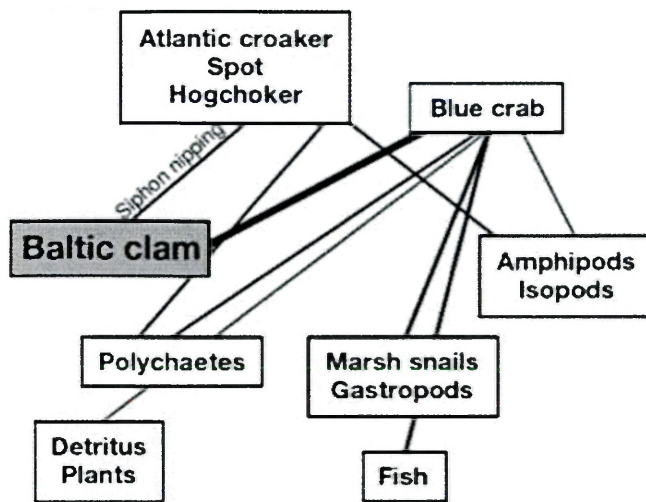
Holly Cove Predators		Parameter Estimates ( $\pm$ SE)					
Response	Adj. R <sup>2</sup>	$\beta_0$ Intercept	$\beta_1$ Year (2011)	$\beta_2$ Treat C2	$\beta_3$ Treat I	$\beta_4$ Y×T (C2)	$\beta_5$ Y×T (I)
Blue Crab Abundance (raw)	0.425	<b>6.0</b> $\pm$ <b>1.72</b>	-0.80 $\pm$ 2.31	-1.5 $\pm$ 2.22	<b>6</b> $\pm$ <b>2.22</b>	-2.45 $\pm$ 3.2	<b>-6.7</b> $\pm$ <b>2.96</b>
Fish Abundance (Box-Cox)	0.340	<b>4.69</b> $\pm$ <b>0.32</b>	0.52 $\pm$ 0.51	-0.58 $\pm$ 0.41	-0.86 $\pm$ 0.43	<i>-1.16</i> $\pm$ <i>0.69</i>	-0.28 $\pm$ 0.67
Fish Diversity (logit)	-0.192	-0.17 $\pm$ 0.38	0.02 $\pm$ 0.60	0.40 $\pm$ 0.48	0.55 $\pm$ 0.50	0.20 $\pm$ 0.82	-0.19 $\pm$ 0.79

**Table 21.** Parameter estimates for Windy Hill predators from the linear model (similar to  $g_3$  for 3-mm infauna). Parameter estimates with 95% confidence intervals that do not include 0 (estimates significant at  $\alpha = 0.05$ ) are in bold, and with 80% confidence intervals (for interaction estimates only) that do not include 0 ( $\alpha = 0.2$ ) in italics. Xs are parameters not included in the model. Year = 2011. Treat = shoreline treatment (C2 = control 2, I = impact). Y×T = year and treatment interaction for the shoreline treatment in parentheses.

Windy Hill Predators		Parameter Estimates ( $\pm$ SE)					
Response	Adj. R <sup>2</sup>	$\beta_0$ Intercept	$\beta_1$ Year (2011)	$\beta_2$ Treat C2	$\beta_3$ Treat I	$\beta_4$ Y×T (C2)	$\beta_5$ Y×T (I)
Blue Crab Abundance (raw)	0.194	2.77E-16 $\pm 0.25$	<b>1.00</b> $\pm$ <b>0.32</b>	0.17 $\pm$ 0.37	0.17 $\pm$ 0.37	-0.67 $\pm$ <i>0.50</i>	<b>-1.17</b> $\pm$ <b>0.50</b>
Fish Abundance (Box-Cox)	0.413	<b>3.40</b> $\pm$ <b>0.32</b>	-0.87 $\pm 0.46$	0.11 $\pm$ 0.46	0.47 $\pm$ 0.46	0.39 $\pm$ 0.64	0.28 $\pm$ 0.62
Fish Diversity (logit)	0.207	-0.28 $\pm$ 0.58	1.04 $\pm$ 0.82	-1.41 $\pm$ 0.82	-0.26 $\pm$ 0.82	2.01 $\pm$ <i>1.16</i>	1.77 $\pm$ <i>1.11</i>

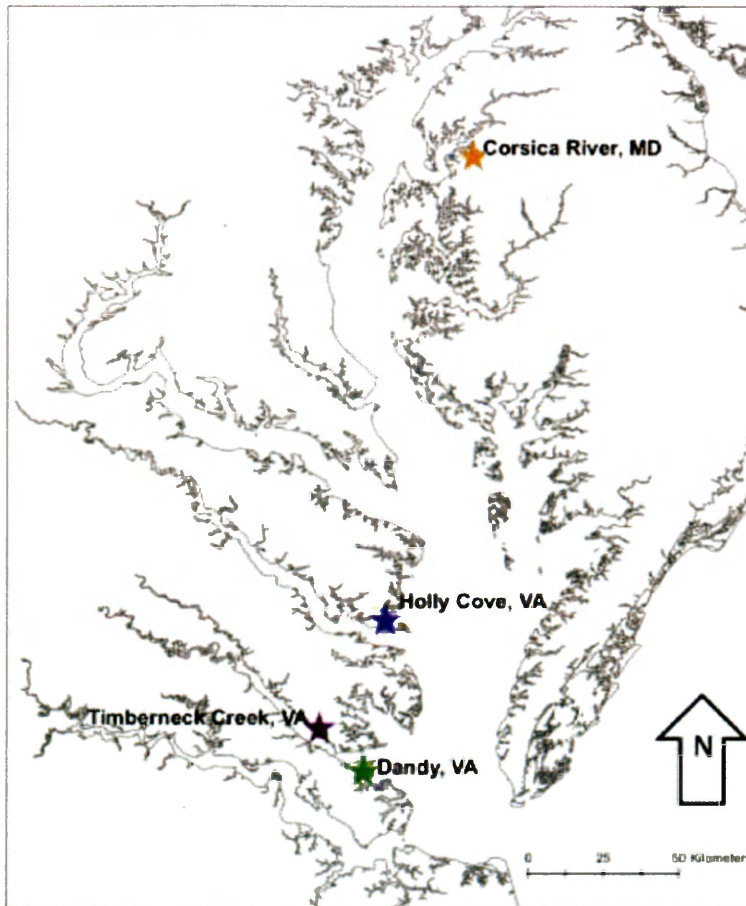
**Table 22.** Summary of significant effects of shoreline modification for all organisms and all response variables at all sites. Significant interactions between year and impact shoreline treatments compared to the reference (control 1 shorelines in 2010) represent significant effects of shoreline modification and are denoted with shaded boxes. Arrows indicate the direction of the shoreline effect; up arrows indicate a positive effect of shoreline modification, down arrows indicate a negative effect. Large arrows (30 pt font) indicate parameter estimates with 95% confidence intervals that do not include 0 (estimates significant at  $\alpha = 0.05$ ). Smaller arrows (16 pt) indicate parameter estimates with 80% confidence intervals that do not include 0 ( $\alpha = 0.2$ ). Response variables are abbreviated as follows: Dens = density, Biom = biomass, Div = diversity, Abund = abundance. Site abbreviations as in Table 1. Xs indicate that there were no significant effects attributable to shoreline modification. N/D = no data, as fish response variables at Dandy were not modeled.

Site	Infauna: 3 mm			Infauna: 500 $\mu$ m			Blue Crabs		Fish	
	Dens	Biom	Div	Dens	Biom	Div	Abund	Div	Abund	Div
TIM	x	<b>↑ I1 (Docks)</b>	<b>↓ I1 (Docks)</b>	x	x	x	<b>↑ I1 (Docks)</b> <b>↑ I2 (Riprap)</b>	x	x	x
DAN	x	x	x	x	<b>↑ (Riprap)</b>	x	x		N/D	N/D
HOL	x	x	x	<b>↑ (Riprap)</b>	x	x	<b>↓ (Riprap)</b>	x	x	x
WIN	x	x	<b>↓ (LS)</b>	x	<b>↓ (LS)</b>	x	x		x	x

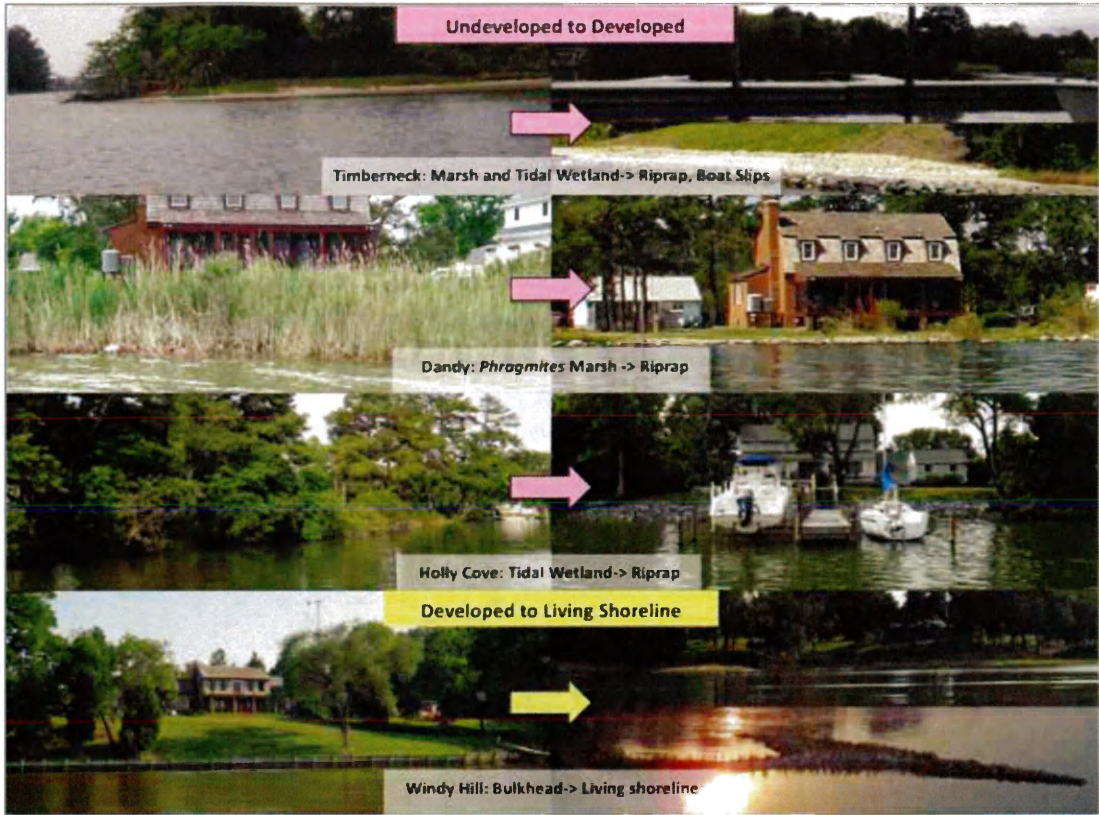


**Figure 1.** Simplified Chesapeake Bay food web in soft sediment benthic systems showing important linkages for the Baltic clam, *M. balthica*. The bold line between the Baltic clam and the Blue crab emphasizes the importance of this trophic link. Adapted from Lipcius et al. 2007.

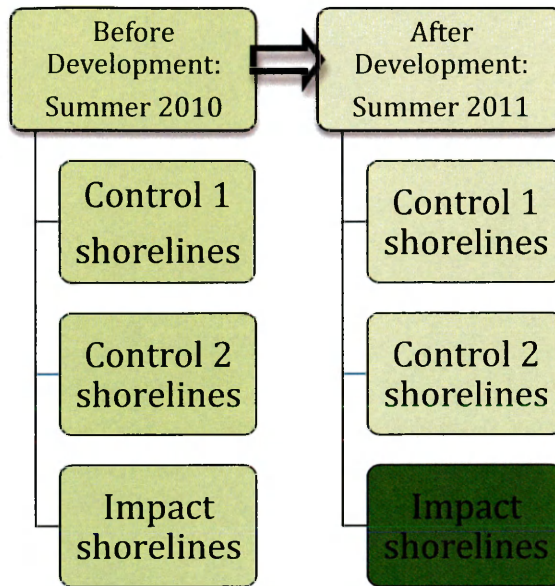
BACI sampling locations, Chesapeake Bay, USA



**Figure 2.** Map of Chesapeake Bay showing general locations of study sites within three subestuaries of Chesapeake Bay. Windy Hill is located on the Corsica River, which is a tributary of the Chester River in Maryland; Holly Cove is located in Tabbs Creek, a tributary of Fleets Bay just north of the Rappahannock River in Virginia; Timberneck Creek is a tributary of the York River, and Dandy is located at the mouth of the York River in Virginia. Map created by Theresa Davenport.

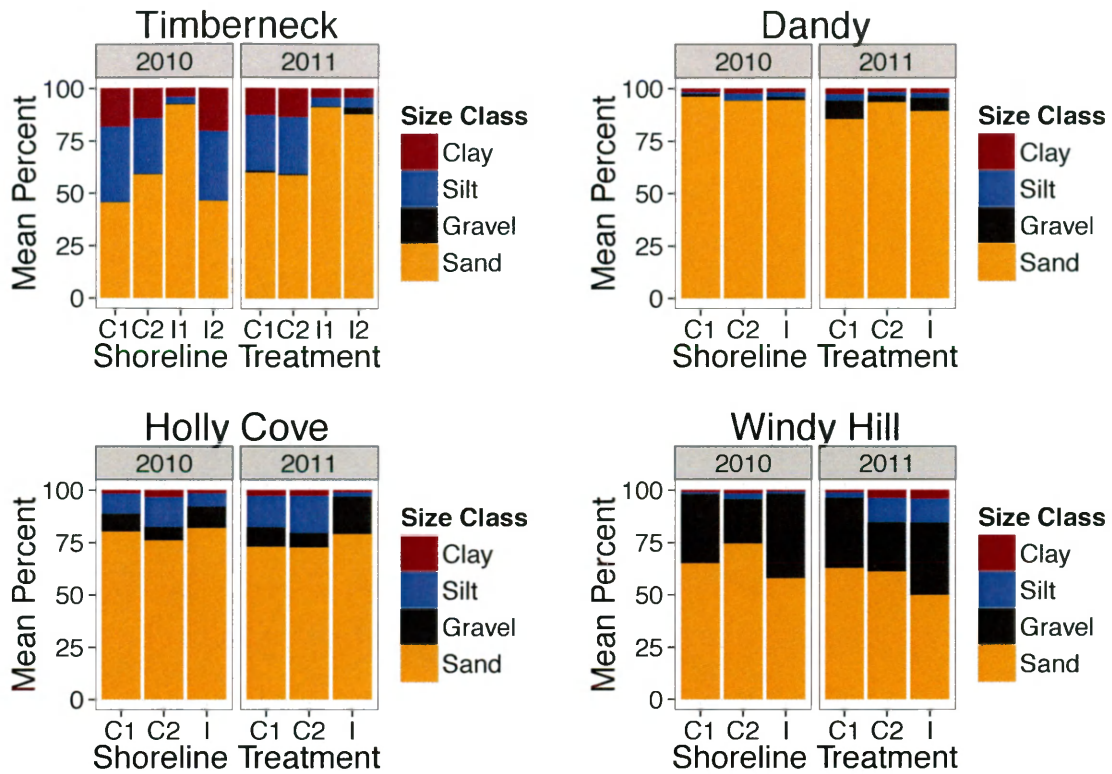


**Figure 3.** Photos of impact shorelines at BACI project sites before (left panels) and after (right panels) shoreline modification.

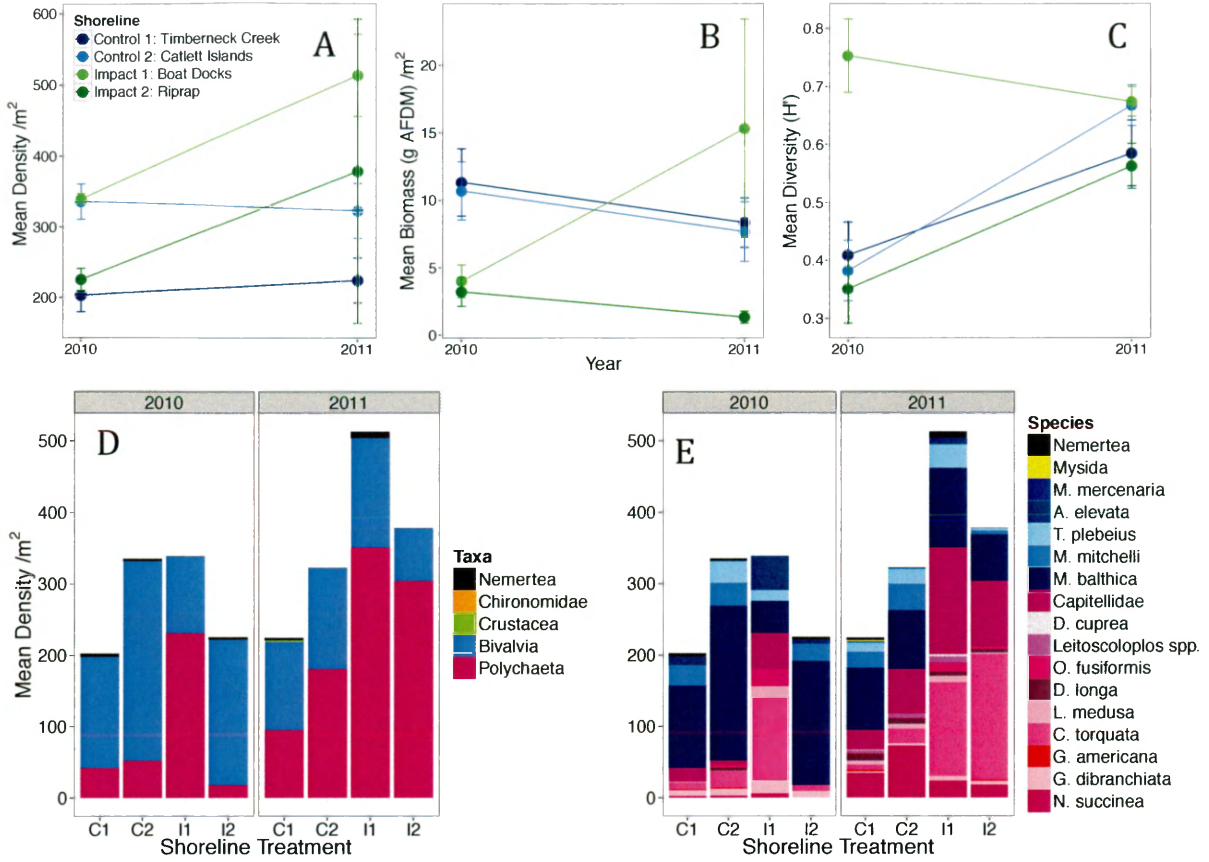


**Figure 4.** Schematic diagram of sampling treatments and differences between the treatments (as seen in different color shades) for the BACI study at each location. The box shading indicates the high level of similarity between all sites in 2010, and the control site in 2011, with a bigger dissimilarity expected at impact sites in 2011.

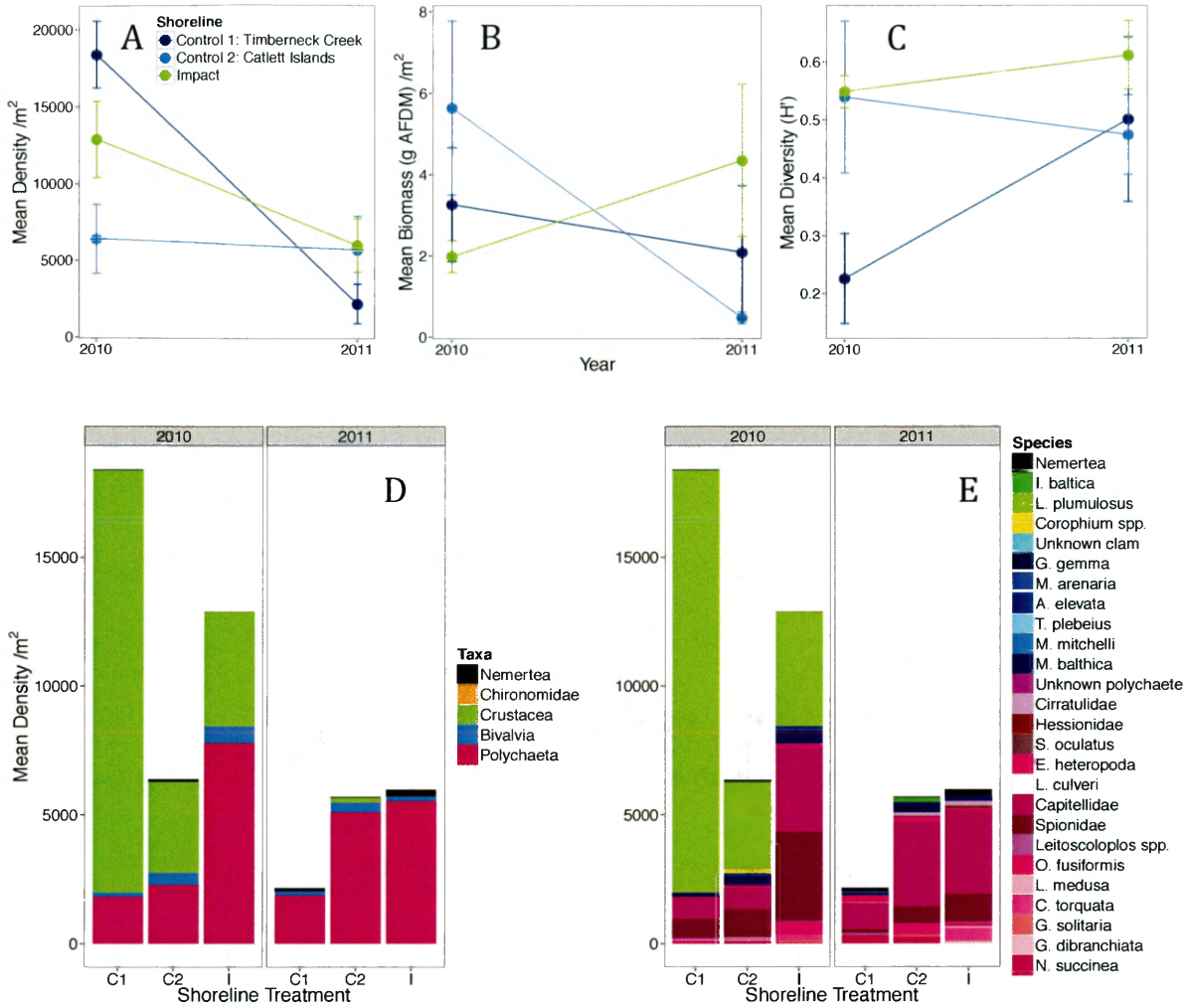




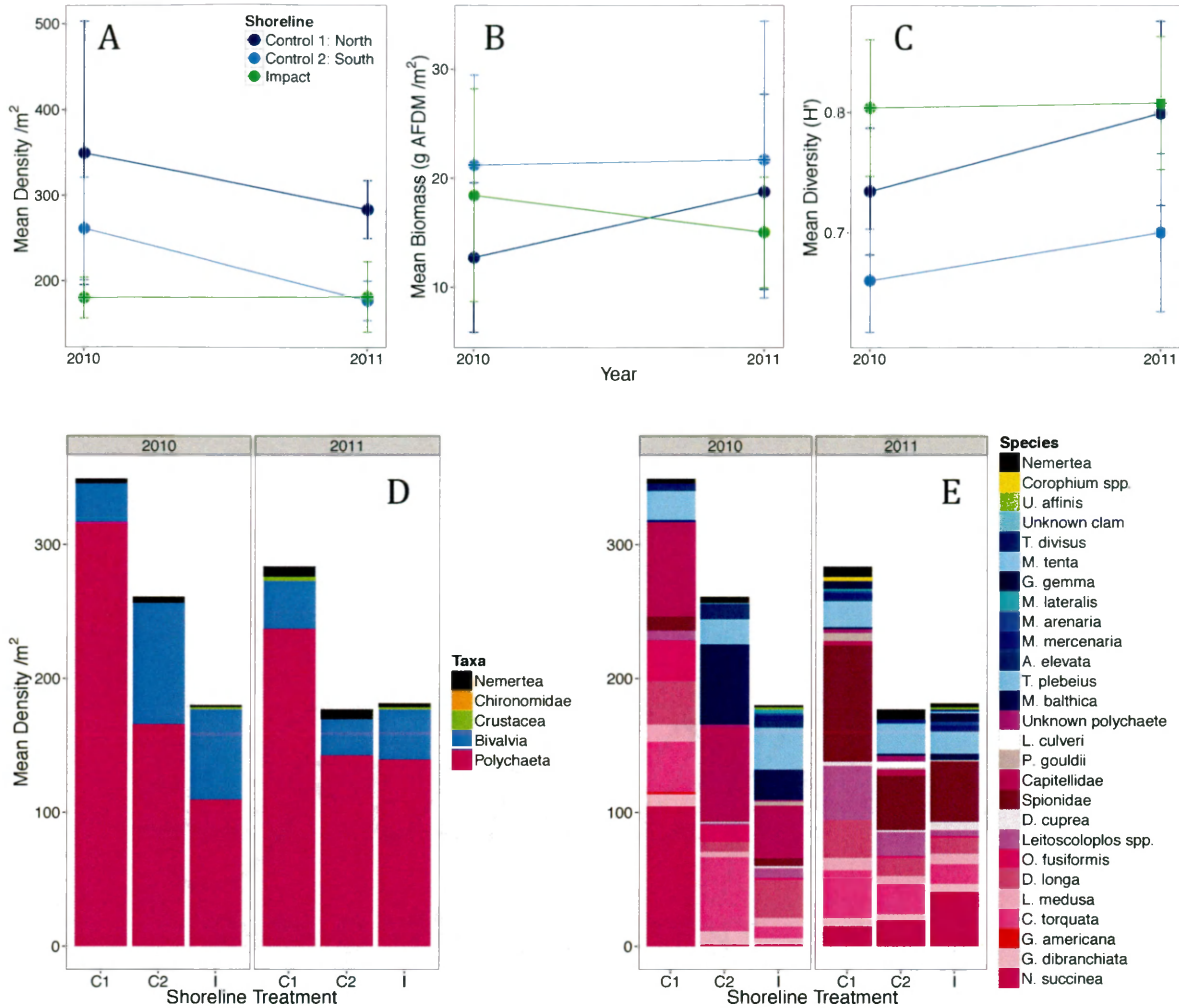
**Figure 5.** Effect of year and shoreline treatment on mean sediment grain size as represented by mean percent of four size classes: sand, gravel, silt, and clay. Gravel fractions include detritus. Site names are labeled at the top of each plot. Sample size as in table 4, infauna.



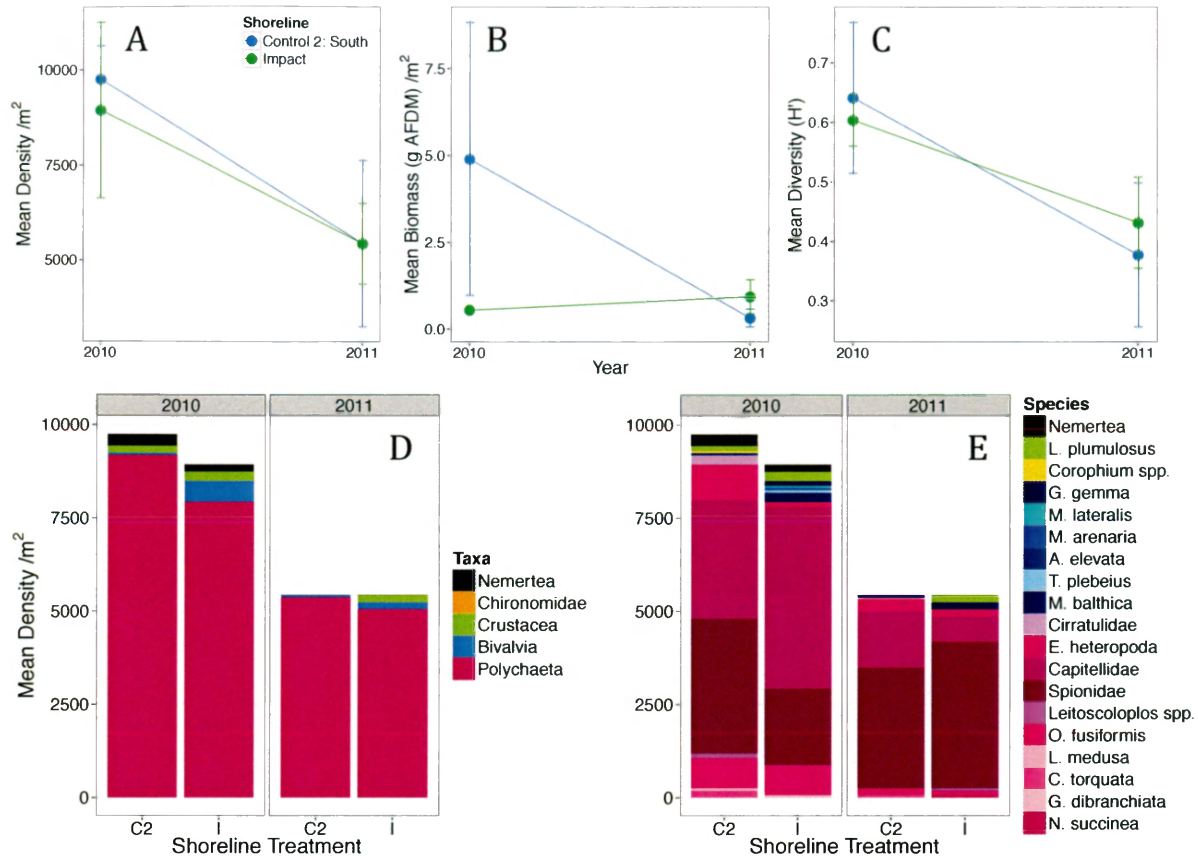
**Figure 6 A-E.** Effect of year and shoreline treatment on 3-mm infauna at Timberneck, measured by A) mean density per m<sup>2</sup> (top left), B) mean biomass (g ash-free dry mass) per m<sup>2</sup> (top center), C) mean Shannon diversity, H' log<sub>10</sub> (top right) D) community composition by the most common taxa (bottom left), and E) community composition to the lowest possible taxonomic classification (bottom right). Error bars are 1 SEM, and sample sizes per year and treatment combination as in Table 3. For error bars of plots D and E, see plot A. Shoreline treatments as follows: C1 = control 1: Timberneck Creek, C2 = control 2: Catlett Islands, I1 = impact 1: boat docks, I2 = impact 2: riprap.



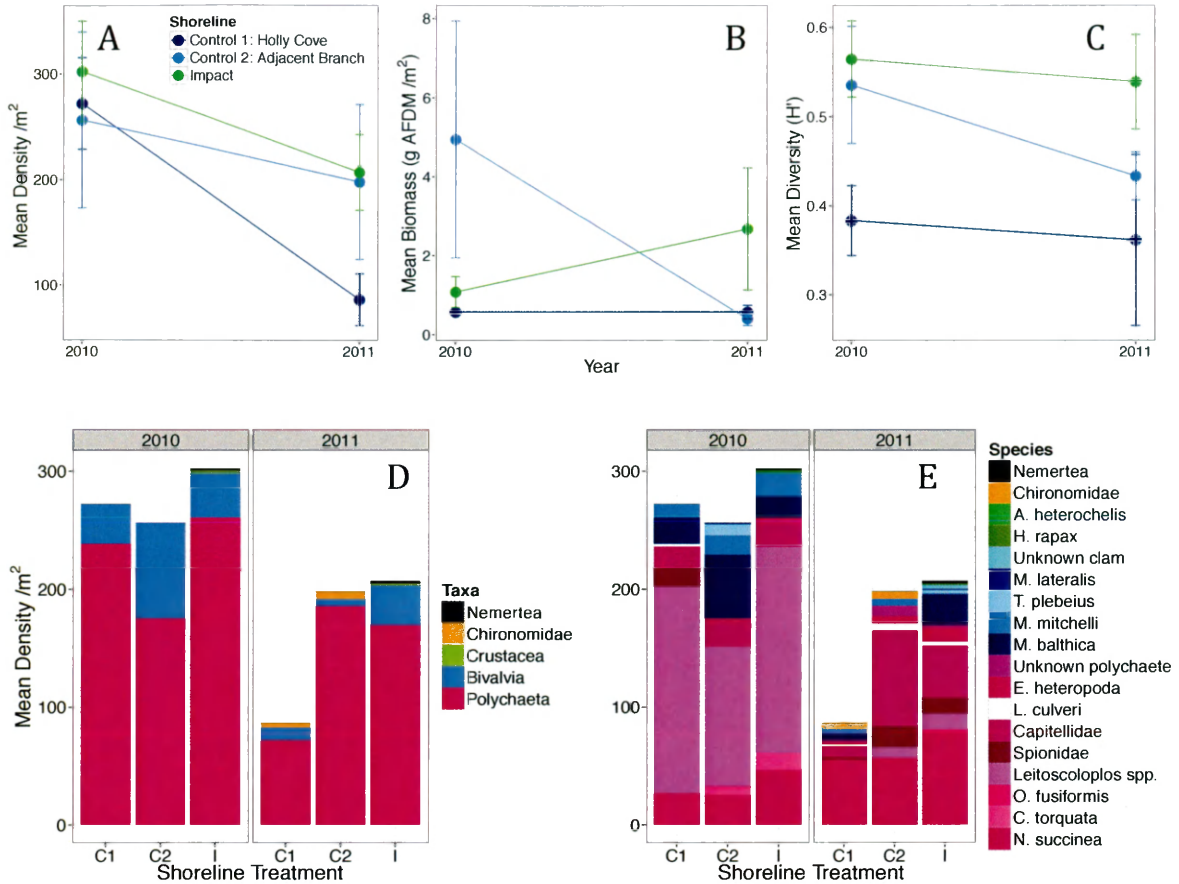
**Figure 7 A-E.** Effect of year and shoreline treatment on 500- $\mu$ m infauna at Timberneck, measured by A) mean density per  $m^2$  (top left), B) mean biomass (g ash-free dry mass) per  $m^2$  (top center), C) mean Shannon diversity,  $H'$   $\log_{10}$  (top right) D) community composition by the most common taxa (bottom left), and E) community composition to the lowest possible taxonomic classification (bottom right). Error bars are 1 SEM, and sample sizes per year and treatment combination as in Table 3. For error bars of plots D and E, see plot A. Shoreline treatments as in Figure 6.



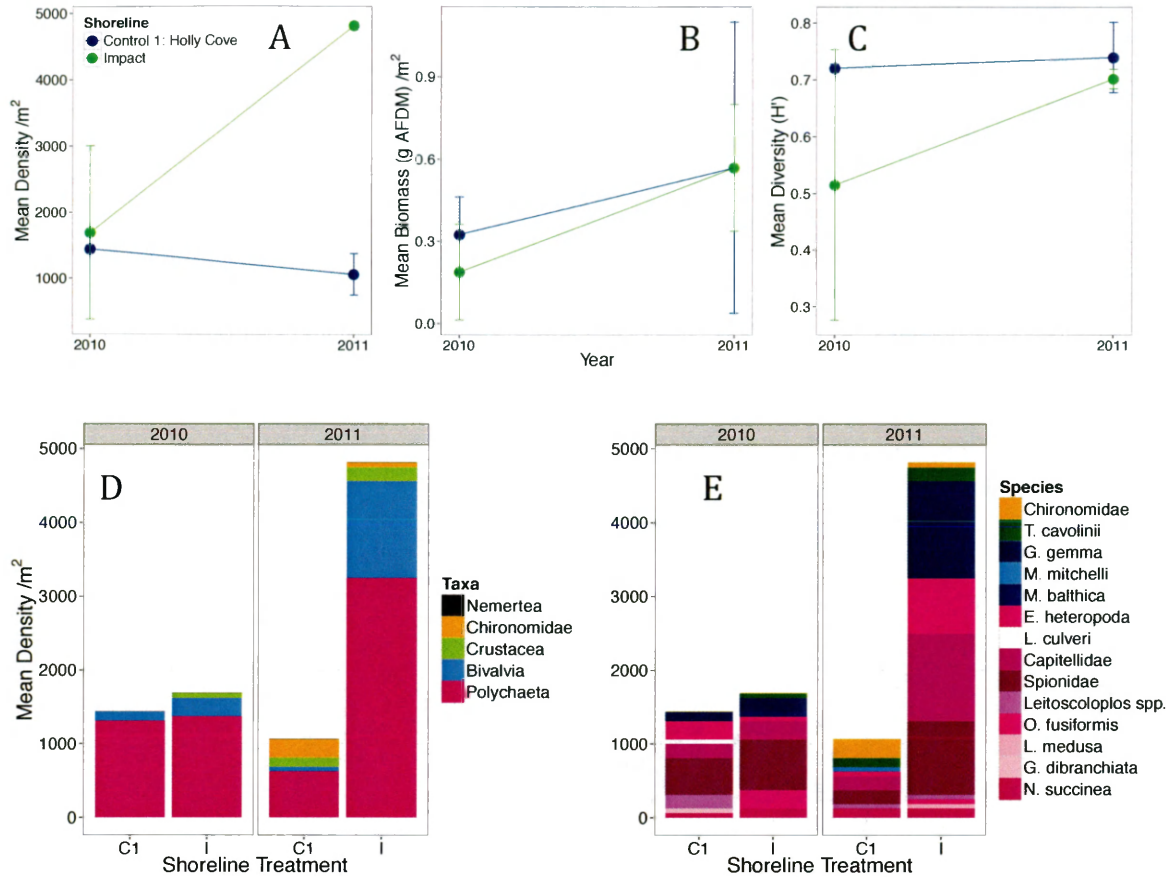
**Figure 8A-E.** Effect of year and shoreline treatment on 3-mm infauna at Dandy, measured by A) mean density per  $m^2$  (top left), B) mean biomass (g ash-free dry mass) per  $m^2$  (top center), C) mean Shannon diversity,  $H' \log_{10}$  (top right) D) community composition by the most common taxa (bottom left), and E) community composition to the lowest possible taxonomic classification (bottom right). Error bars are 1 SEM, and sample sizes per year and treatment combination as in Table 3. For error bars of plots D and E, see plot A. Shoreline treatments as follows: C1 = control 1: marsh shoreline north of impact, C2 = control 2: marsh shoreline south of impact, I = impact: riprap.



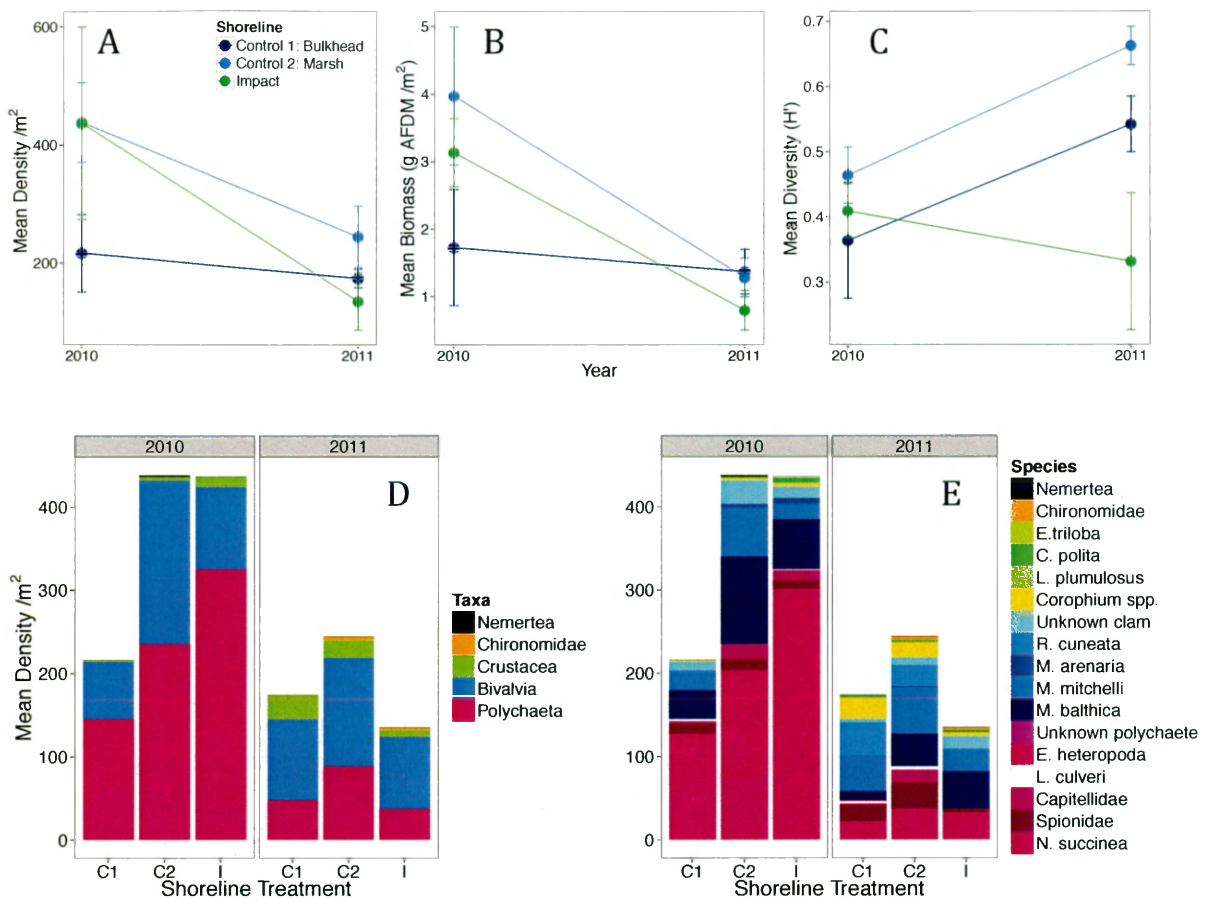
**Figure 9A-E.** Effect of year and shoreline treatment on 500- $\mu$ m infauna at Dandy, measured by A) mean density per  $m^2$  (top left), B) mean biomass (g ash-free dry mass) per  $m^2$  (top center), C) mean Shannon diversity,  $H' \log_{10}$  (top right) D) community composition by the most common taxa (bottom left), and E) community composition to the lowest possible taxonomic classification (bottom right). Error bars are 1 SEM, and sample sizes per year and treatment combination as in Table 3. For error bars of plots D and E, see plot A. Shoreline treatments as in Figure 8.



**Figure 10 A-E.** Effect of year and shoreline treatment on 3-mm infauna at Holly Cove, measured by A) mean density per m<sup>2</sup> (top left), B) mean biomass (g ash-free dry mass) per m<sup>2</sup> (top center), C) mean Shannon diversity, H' log<sub>10</sub> (top right) D) community composition by the most common taxa (bottom left), and E) community composition to the lowest possible taxonomic classification (bottom right). Error bars are 1 SEM, and sample sizes per year and treatment combination as in Table 3. For error bars of plots D and E, see plot A. Shoreline treatments as follows: C1 = control 1: tidal wetland adjacent to impact at Holly Cove, C2 = control 2: tidal wetland in adjacent branch, I = impact: riprap.

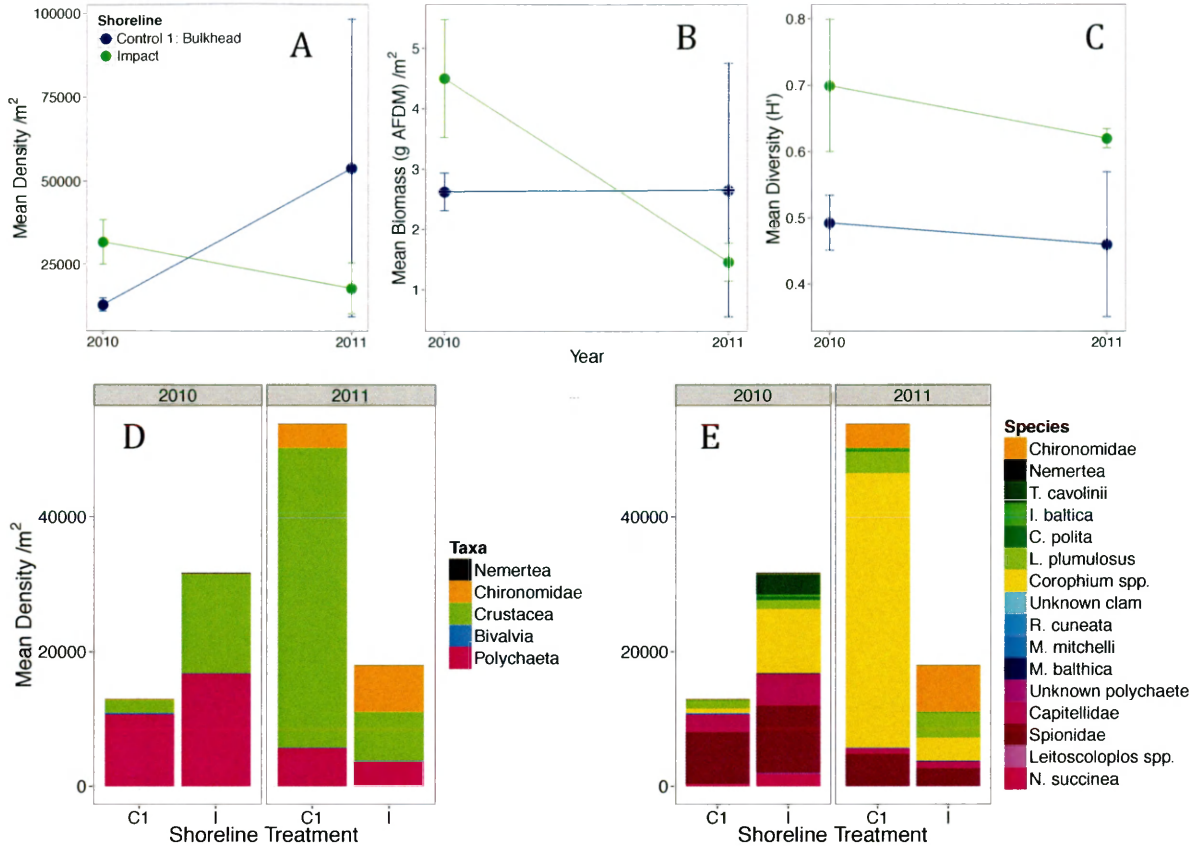


**Figure 11A-E.** Effect of year and shoreline treatment on 500- $\mu$ m infauna at Holly Cove, measured by A) mean density per m<sup>2</sup> (top left), B) mean biomass (g ash-free dry mass) per m<sup>2</sup> (top center), C) mean Shannon diversity, H' log<sub>10</sub> (top right) D) community composition by the most common taxa (bottom left), and E) community composition to the lowest possible taxonomic classification (bottom right). Error bars are 1 SEM, and sample sizes per year and treatment combination as in Table 3. For error bars of plots D and E, see plot A. Shoreline treatments as in Figure 10.

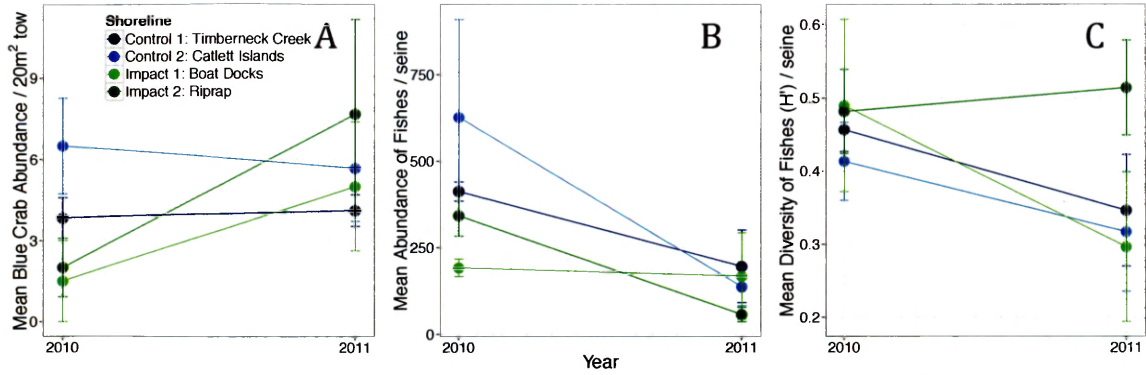


**Figure 12 A-E.** Effect of year and shoreline treatment on 3-mm infauna at Windy Hill, measured by A) mean density per m<sup>2</sup> (top left), B) mean biomass (g ash-free dry mass) per m<sup>2</sup> (top center), C) mean Shannon diversity, H' log<sub>10</sub> (top right) D) community composition by the most common taxa (bottom left), and E) community composition to the lowest possible taxonomic classification (bottom right). Error bars are 1 SEM, and sample sizes per year and treatment combination as in Table 3. For error bars of plots D and E, see plot A. Shoreline treatments as follows: C1 = control 1: bulkhead shorelines, C2 = marsh shorelines, I = impact: living shoreline.

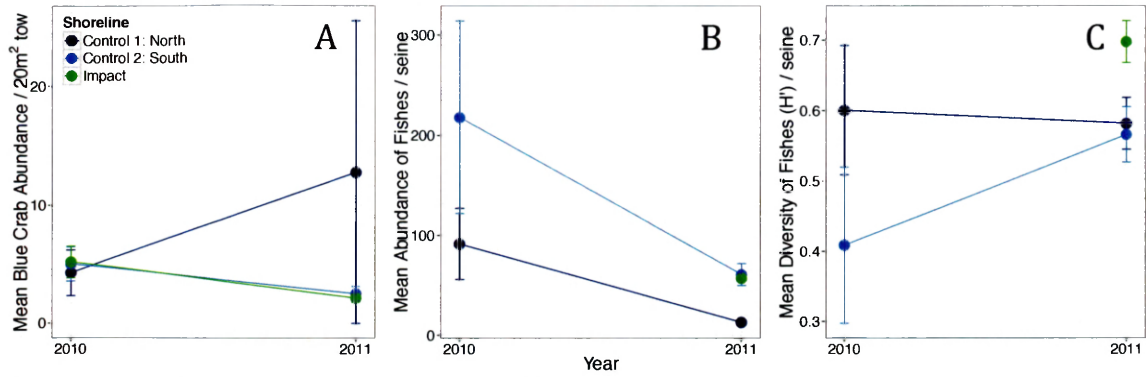




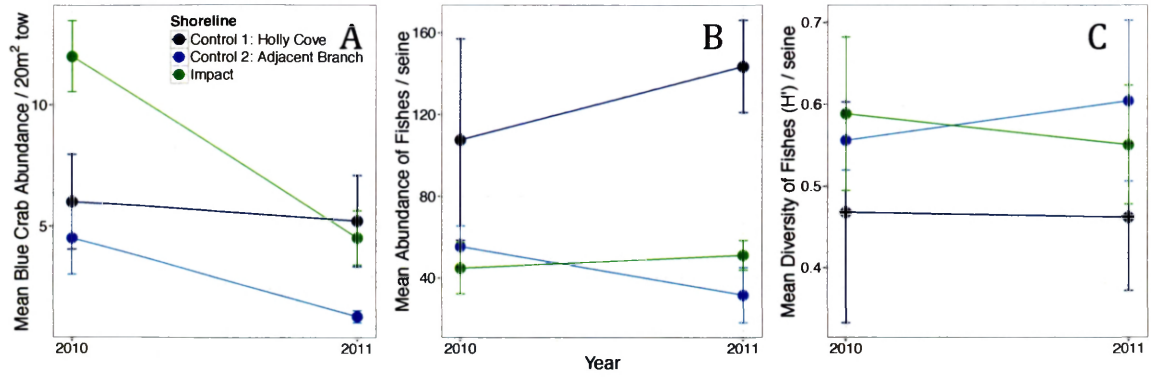
**Figure 13 A-E.** Effect of year and shoreline treatment on 500- $\mu$ m infauna at Holly Cove, measured by A) mean density per  $m^2$  (top left), B) mean biomass (g ash-free dry mass) per  $m^2$  (top center), C) mean Shannon diversity,  $H'$   $\log_{10}$  (top right) D) community composition by the most common taxa (bottom left), and E) community composition to the lowest possible taxonomic classification (bottom right). Error bars are 1 SEM, and sample sizes per year and treatment combination as in Table 3. For error bars of plots D and E, see plot A. Shoreline treatments as in Figure 12.



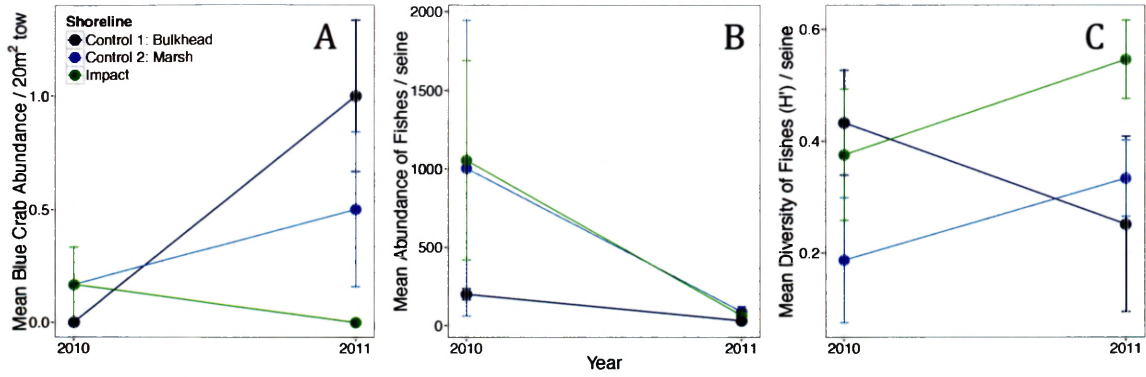
**Figure 14 A-C.** Effect of year and shoreline treatment on mean abundance of blue crabs per 20 m<sup>2</sup> tow (far left), abundance of fishes per 182.4 m<sup>2</sup> seine (center), and Shannon diversity, (H' log<sub>10</sub>) of fishes per seine (far right) at Timberneck. Error bars are 1 SEM, and sample size as in Table 3.



**Figure 15 A-C.** Effect of year and shoreline treatment on mean abundance of blue crabs per 20 m<sup>2</sup> tow (far left), fishes per 182.4 m<sup>2</sup> seine (center), and Shannon diversity, ( $H'$   $\log_{10}$ ) of fishes per seine (far right) at Dandy. Error bars are 1 SEM, and sample size as in Table 3. Note that no seines for fishes were conducted in 2010 at impact sites (center and right plots).



**Figure 16 A-C.** Effect of year and shoreline treatment on mean abundance of blue crabs per 20 m<sup>2</sup> tow (far left), fishes per 182.4 m<sup>2</sup> seine (center), and Shannon diversity, (H' log<sub>10</sub>) of fishes per seine (far right) at Holly Cove. Error bars are 1 SEM, and sample size as in Table 3.



**Figure 17 A-C.** Effect of year and shoreline treatment on mean abundance of blue crabs per 20 m<sup>2</sup> tow (far left), fishes per 182.4 m<sup>2</sup> seine (center), and Shannon diversity, ( $H'$   $\log_{10}$ ) of fishes per seine (far right) at Windy Hill. Error bars are 1 SEM, and sample size as in Table 3.

## VITA

Born in Doylestown, Pennsylvania, August 3, 1986. Graduated from Pennridge High School, Perkasio, Pennsylvania in 2004. Earned a B.S. in Biology and a B.S. in Environmental Studies, *summa cum laude*, from Gettysburg College in 2008. Worked as a laboratory assistant at Georgia Institute of Technology from 2008-2009. Entered the Master's program in the School of Marine Science, College of William and Mary in August 2009.