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Diann J. Prosser

Thomas E Jordan

Jessica L. Nagel

Rochelle D. Seitz
*Virginia Institute of Marine Science*

Donald E. Weller

*See next page for additional authors*

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## Authors

Diann J. Prosser, Thomas E Jordan, Jessica L. Nagel, Rochelle D. Seitz, Donald E. Weller, and Dennis F. Whigham

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# Impacts of Coastal Land Use and Shoreline Armoring on Estuarine Ecosystems: an Introduction to a Special Issue

Diann J. Prosser[1] · Thomas E. Jordan[2] · Jessica L. Nagel[1] · Rochelle D. Seitz[3] ·
Donald E. Weller[2] · Dennis F. Whigham[2]

**Abstract** The nearshore land-water interface is an important ecological zone that faces anthropogenic pressure from development in coastal regions throughout the world. Coastal waters and estuaries like Chesapeake Bay receive and process land discharges loaded with anthropogenic nutrients and other pollutants that cause eutrophication, hypoxia, and other damage to shallow-water ecosystems. In addition, shorelines are increasingly armored with bulkhead (seawall), riprap, and other structures to protect human infrastructure against the threats of sea-level rise, storm surge, and erosion. Armoring can further influence estuarine and nearshore marine ecosystem functions by degrading water quality, spreading invasive species, and destroying ecologically valuable habitat. These detrimental effects on ecosystem function have ramifications for ecologically and economically important flora and fauna. This special issue of *Estuaries and Coasts* explores the interacting effects of coastal land use and shoreline armoring on estuarine and coastal marine ecosystems. The majority of papers focus on the Chesapeake Bay region, USA, where 50 major tributaries and an extensive watershed ($\sim 167{,}000~\mathrm{km}^2$), provide an ideal model to examine the impacts of human activities at scales ranging from the local shoreline to the entire watershed. The papers consider the influence of watershed land use and natural versus armored shorelines on ecosystem properties and processes as well as on key natural resources.

Shallow estuarine waters are biologically active interfaces between the land and the sea that provide many beneficial ecosystem services for diverse aquatic and terrestrial animal communities (Costanza et al. 1997; Orth et al. 2006; Barbier et al. 2011). Shallow nearshore habitats, including non-vegetated shallows, tidal wetlands, and submerged aquatic vegetation (SAV), serve as critical feeding and nursery habitats for many fisheries species (Ruiz et al. 1993; Beck et al. 2001; King et al. 2005; Seitz et al. 2014; Sheaves et al. 2015) as well as important seasonal habitats for migratory species (Erwin 1996). These habitat functions enhance energy flow to higher trophic levels (Dittel et al. 1995; Miller et al. 1996; Clark et al. 2003; Heck et al. 2008).

Nearshore habitats are being degraded by multiple stressors. Shoreline armoring, land development, and invasive species threaten the economically and ecologically important species that depend on these habitats. Here, we focus on temperate estuaries and the effects of human land use and shoreline armoring on the interacting biotic components of the land-water interface, including tidal wetlands, SAV, and fauna, such as benthic invertebrates, epibenthic crustaceans, fish, and waterbirds. These biotic components are influenced by the physical properties of the shoreline and by water quality, which, in turn, is influenced by inputs from the watershed and exchanges with adjacent estuarine water.

✉ Diann J. Prosser
  dprosser@usgs.gov

[1] Patuxent Wildlife Research Center, Beltsville Laboratory, U.S. Geological Survey, 10300 Baltimore Avenue, Beltsville, MD 20705, USA

[2] Smithsonian Environmental Research Center, 647 Contees Wharf Road, Edgewater, MD 21037, USA

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

The expansion of coastal human development and armored shorelines throughout the world (Ache et al. 2013; Gittman et al. 2015) creates a growing need for information on the interacting effects of land use and shoreline armoring on nearshore ecosystems.

The world's population is concentrated near coastlines, and that concentration is increasing (Vitousek et al. 1997; Small and Nicholls 2003; Ache et al. 2013; Paerl et al. 2014). Human activities in the nearshore zone can have significant negative impacts on water quality and nearshore habitats. Runoff of nitrogen (N) and phosphorus (P) from urban and agricultural watersheds (e.g., Jordan et al. 2003; Weller et al. 2003) threatens estuarine habitats worldwide through increased algal production, decreased water clarity, and increased occurrence of hypoxia (Doney 2010; Howarth et al. 2011). One major consequence of decreased water quality has been the decline in SAV and associated fauna (Bologna and Heck 1999; Orth et al. 2006; Waycott et al. 2009; Brady and Targett 2013). Nutrient enrichment also impacts coastal wetlands by enhancing the spread of invasive species, such as the Common Reed *Phragmites australis* (hereafter *Phragmites*) (Silliman and Bertness 2004; Kettenring et al. 2015), or by reducing root production, which negatively impacts wetland accretion (Langley et al. 2009) and the integrity of intertidal wetland banks (Deegan et al. 2012).

Shoreline alteration, especially armoring, is another important aspect of land use that affects shallow-water coastal systems. The threats of erosion and property loss from storm surge and sea-level rise motivate land owners to install hardened structures to preserve their properties (Dugan et al. 2011; Gittman et al. 2015). These include vertical seawalls or bulkheads (wooden, concrete, or vinyl walls); riprap revetments (granite or concrete stones or boulders placed at an angle); or other structures, such as piers, breakwaters, or low-crested coastal defense structures (Fig. 1). Some coastal cities have 50% or more of their shorelines armored (Patrick et al. 2014; Gittman et al. 2016b).

There are unintended consequences of shoreline armoring for natural communities. Natural vegetation is replaced resulting in the loss of important habitat services, such as provision of refuge for fauna (Duhring 2008) (Fig. 2). Secondary effects of armoring differ with the type of structure installed. For example, bulkheads reduce filtration of upland run-off by severing the land-water interface (Currin et al. 2010) and may also be sites where enhanced wave reflection erodes sediment and deepens nearshore habitats (NRC 2007; Toft et al. 2007) (Fig. 2). On the other hand, riprap may allow limited exchange of nutrients with the upland and also provide habitat for some biota (Seitz et al. 2006; Davis et al. 2008; Bilkovic and Mitchell 2013; Balouskus and Targett 2016). The combination of coastal erosion and upland development results in "coastal squeeze," whereby low-lying, intertidal

areas that would normally retreat inland in the face of sea-level rise are reduced because man-made structures block that retreat (Doody 2004; Pontee 2013; Kirwan et al. 2016). Detrimental effects of shoreline armoring for coastal ecosystems worldwide are becoming well known (Bulleri 2006; Airoldi and Beck 2007; Gittman et al. 2016b) and can include loss of biodiversity and production (Seitz et al. 2006; Bilkovic and Roggero 2008; Dugan et al. 2008; Peterson and Lowe 2009).

Many coastal states in the USA now have incentives or requirements to install "living shorelines" instead of traditional armored structures. Living shorelines incorporate natural elements such as wetland plantings, coral reefs, or shellfish reefs along the shoreline (Temmerman et al. 2013). The natural elements can also be combined with structures like a riprap sill, coir fiber logs with sand fill, oyster shell bags, or shellfish reef just offshore (Figs. 1f and 2) (Duhring 2008). Living shorelines have become increasingly popular because they perform as well as bulkhead or riprap at controlling erosion (Currin et al. 2010; Gittman et al. 2014, 2016a). Moreover, the added natural elements increase habitat value for fauna, maintain nearshore depth and sediment type, and sequester carbon while improving protection from erosion (Davis et al. 2015; Bilkovic et al. 2016). Local hydrodynamics, tidal range, elevation, shoreline geomorphology, and sediment characteristics can all influence the living shoreline design at a specific site (Roberts 2008; Currin et al. 2015; Sutton-Grier et al. 2015). While innovations to improve both the erosion protection and the ecological benefits of living shoreline designs are ongoing, further research and long-term monitoring of the effectiveness of these structures for protecting shorelines will be essential for informing future management decisions (Bilkovic et al. 2016).

## Focus of the Special Issue

This special issue considers the interacting impacts of coastal land use and shoreline armoring on estuarine and coastal marine ecosystems. We synthesize recent research to better understand the interacting effects of shoreline armoring and watershed land use on estuarine resources and processes. The 13 research papers address the effects of different watershed land uses (development, agricultural, forested) and different man-made shoreline types (bulkhead, riprap) compared to natural shorelines (beaches, natural wetlands, wetlands dominated by the invasive species *Phragmites*) on ecosystem properties (water quality, hydrodynamics, nutrient dynamics, and biogeochemistry) and on key natural resources (SAV, wetlands, benthic communities, macroinvertebrates, fishes, and waterbirds). The final paper presents our efforts to integrate research and management.

**Fig. 1** Examples of shorelines dominated by **a** bulkhead, **b** riprap revetment, **c** beach, **d** natural wetland, **e** *Phragmites australis* wetland, and **f** living shoreline. Photo of the living shoreline was taken 5 years after construction. The poles are the remnants of the temporary fencing installed to protect the wetland plants from goose predation after planting and to deter boats from landing on the beach



a) **Bulkhead**

b) **Riprap**

c) **Beach**

d) **Natural Wetland**

e) *Phragmites* **Wetland**

f) **Living Shoreline**

All but two of the papers result from a 7-year, multi-investigator, interdisciplinary research collaboration among 19 principal investigators from eight institutions. The project evaluated the physical and ecological influences of watershed land use and shoreline development along the entire salinity gradient of the largest bay in the USA, Chesapeake Bay, and in ocean embayments along the mid-Atlantic coast. The research used a comparative approach exploiting the many subestuaries in this region. The Chesapeake Bay (Fig. 3) has 50 major tributaries and a 167,000 km² watershed, providing an ideal model to examine the impacts of human activities at scales ranging from the entire system to local alterations of land use or shorelines (the focal scale for many contributions in this volume). Each subestuary, or embayment with at least one perennial tributary, has its own local watershed (Li et al. 2007). Subestuaries differ widely in their watershed land uses and occur across the full range of salinity (Fig. 3), making them convenient, replicated study units for comparing systems dominated by different land uses and salinity regimes (e.g., Li et al. 2007;

Patrick et al. 2014). Subestuaries can also be replicate units for analyzing the effects of other stressors on estuarine responses. For example, subestuary comparisons have been applied to studies of SAV (Li et al. 2007; Patrick et al. 2014), blue crab and bivalve abundance (King et al. 2005), polychlorinated biphenyl (PCB) contamination of fish (King et al. 2004), bird community diversity (DeLuca et al. 2004), and the invasion of wetlands by a non-native form of *Phragmites* (King et al. 2007). The two papers that were not part of the interdisciplinary project (Dugan et al. 2017; Gehman et al. 2017) were chosen to complement the other papers.

## Physical Effects of Armoring

Armoring strongly modifies the interactions of waves with shoreline habitats, causing changes in physical structure and sediment composition; and these effects depend on the type of armoring, wind and tidal patterns, and the type of sediment substrate. While breakwaters can reduce physical energy at the
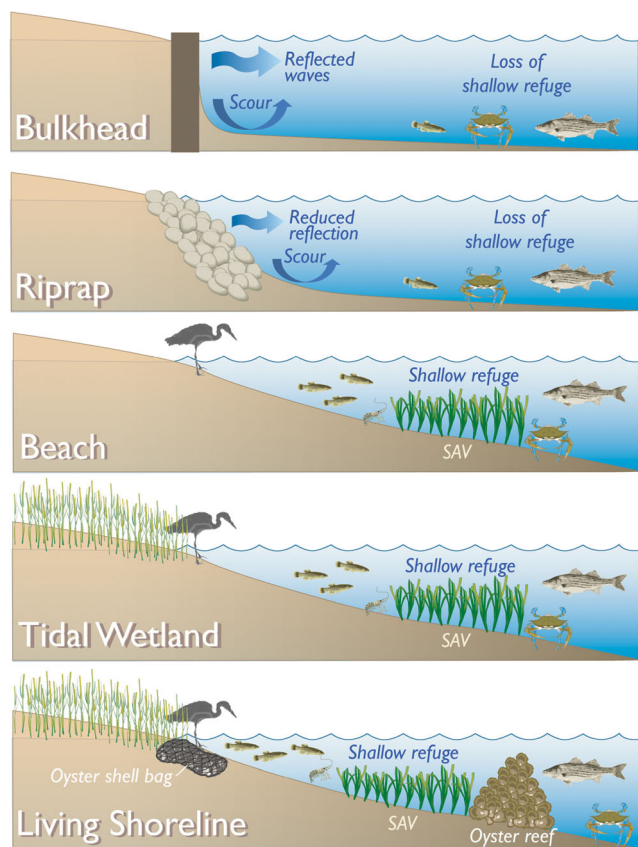
**Fig. 2** Potential differences in habitat value among armored, natural, and living shorelines. Natural shorelines (beach and tidal wetlands) and adjacent submerged aquatic vegetation (SAV) beds provide resources and refuge for many estuarine species. Armoring shorelines with bulkhead or riprap may destroy these habitats and the associated shallow-water refuge. A living shoreline incorporates natural elements such as wetland plantings reinforced with oyster shell bags along the shoreline and a shellfish reef just offshore

shoreline, bulkheads and riprap can increase energy at the shoreline by reflecting waves (Plant and Griggs 1992; Miles et al. 2001; Strayer and Findlay 2010) (Fig. 2). Increased energy can, in turn, increase nearshore erosion and sediment export, which deepen the adjacent littoral zone (Dugan et al. 2011). Armored shorelines can also block supplies of fine or organic particles and new terrestrial material to the nearshore zone (Pilkey and Wright 1988; Griggs 2005; Nordstrom 2014). This can result in coarser sediments (Jennings et al. 2001; Gabriel and Bodensteiner 2012; Strayer et al. 2012), narrowing of beach zones (Hall and Pilkey 1991; Fletcher et al. 1997; Dugan et al. 2008), and reduced accumulation and retention of marine wrack and terrestrial organic debris (Heerhartz et al. 2014) along armored shores. Ultimately, any of these changes to hydrodynamic and sediment depositional processes may influence the extent and position of nearshore habitats (e.g., sandy beaches, wetlands, and SAV beds) adjacent to armored shorelines (Bulleri and Chapman 2010).

A better understanding of the factors influencing local hydrodynamics and erosion rates is essential for informing decisions about shoreline management (e.g., type and extent of structures) in a given region. In this issue, Sanford and Gao (2017) (*Influences of wave climate and sea level on shoreline erosion rates in the Maryland Chesapeake Bay*) investigated spatial correlations between wave forcing, sea-level fluctuations, and shoreline erosion. Using the Simulating Waves Nearshore (SWAN) model and a parametric wave model from the USEPA Chesapeake Bay Program, they combined wave and sea-level data sets with estimates of historical shoreline erosion rates and shoreline characteristics at two different spatial resolutions to explore the factors affecting erosion. Wave power had the most significant influence on erosion, but many other local factors were also implicated. Wetland shorelines showed a more homogeneous, approximately linear relationship between wave power and erosion rates and also appeared to erode faster than bank shorelines for similar wave powers and bank heights. Derived relationships tended to be scale-dependent and spatially variable, so comprehensive local data sets may be necessary for building detailed local predictive relationships.

## Effects of Land Use on Nearshore Water Quality

Human land use activities have important effects on coastal water quality. Sediments generated by agricultural activities, forest loss, and impervious surfaces are delivered to the coast and reduce water clarity. Nutrient pollution (N and P) from agricultural activities, developed lands, point sources (e.g., sewage treatment plants and industry), and atmospheric deposition all contribute to coastal eutrophication (Nixon 1995; Doney 2010; Howarth et al. 2011; Paerl et al. 2014), a central problem in many coastal estuaries (Nixon 1995; de Jonge et al. 2002; Scavia and Bricker 2006). In some regions, such as the Chesapeake Bay (Boesch et al. 2001; Kemp et al. 2005), federal mandates have been established to reduce high nutrient inputs (USEPA 2010).

Anthropogenic changes in water quality degrade ecological health, reduce the value of commercially important fisheries, and lower the capacity of coastal systems to provide other ecosystem services. Nutrient pollution from agriculture and fossil fuel combustion stimulates the growth of aquatic plants, including phytoplankton, nuisance algae, harmful algal blooms, epiphytes, and invasive macrophytes (Doney 2010; Howarth et al. 2011). Phytoplankton in the water column and epiphytes on leaves reduce the light available for survival and growth of aquatic macrophytes (Orth et al. 2006). Changes in the phytoplankton and macrophyte communities alter aquatic food webs and change the habitats of fish, benthic fauna, shellfish, and birds (Kemp et al. 2005). Phytoplankton decomposition drives hypoxia, which reduces the growth and survival of aquatic fauna (Diaz and Rosenberg 2008). Toxic algal blooms can kill fish and impact human health (Glasgow et al. 1995), and toxic pollutants like mercury

**Fig. 3** The Chesapeake Bay and mid-Atlantic Coastal Bays. The local watersheds of tributary subestuaries (black outlines) are colored by dominant land use and stippled by the salinity zone of the subestuary (TF: tidal fresh, OH: oligohaline, MH: mesohaline, or PH: polyhaline). Delineations of subestuaries and their local watersheds and the categorizations of dominant land uses and salinity zones are described in Li et al. (2007) and Patrick et al. (2014). The inset shows the outline of the Chesapeake watershed (gray line) relative to six states in the eastern USA and Washington DC (red dot)
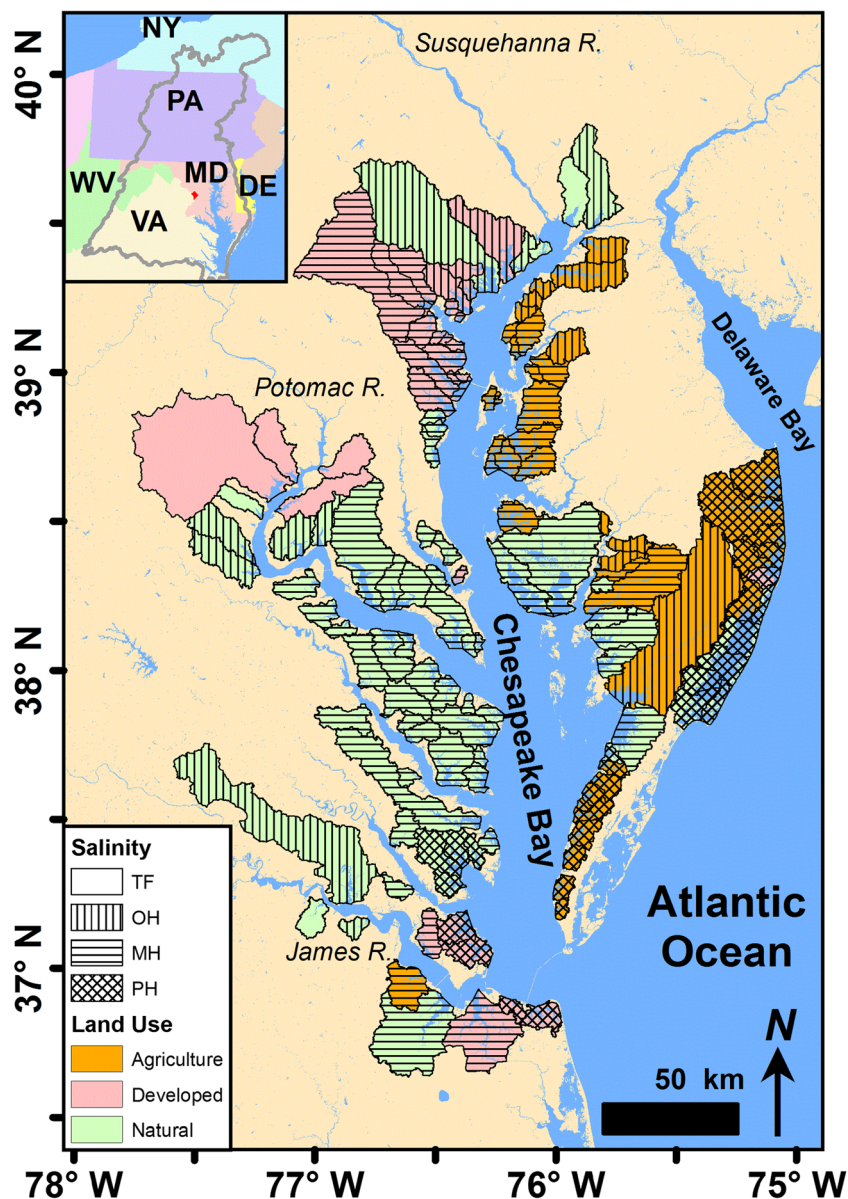
and PCBs contaminate food webs, further stress fauna, and make seafood less suitable for human consumption (Domingo and Bocio 2007).

In this issue, Jordan et al. (2017) (*Effects of local watershed land use on water quality in mid-Atlantic coastal bays and subestuaries of the Chesapeake Bay*) compared water quality at shallow, nearshore locations in 49 Chesapeake subestuaries and mid-Atlantic coastal bays with differing local watershed land use. Concentrations of total N, dissolved ammonium, dissolved inorganic N, and chlorophyll-*a* were positively correlated with the percentages of urban and agricultural land in the watershed. Total phosphorus (TP) and dissolved phosphate also increased with agricultural land but were not affected by urban land use. These relationships indicate the importance of land use in the local watershed and suggest that N is the limiting nutrient for chlorophyll-*a* production in

these systems. In addition, nearshore water quality inside the subestuaries generally differed from that measured in adjacent tidal waters, with TP, organic N, and chlorophyll-*a* concentrations much higher inside the subestuaries, while nitrate concentrations were much higher outside. Therefore, these findings suggest that the importance of local land use and the distinctive biogeochemical environment of nearshore waters should be considered in managing coastal systems.

### Effects of Shoreline Armoring and Land Use on Tidal Wetlands

Tidal wetlands provide many ecosystem services, including carbon sequestration (Drake et al. 2015; Pastore et al. 2017), filtering nutrients and suspended sediments from agricultural or urban runoff (Nixon 1980; Howes et al. 1996; Comin et al.

1997), and supporting diverse faunal communities (Chapman 2003; Parikh and Datye 2003; Scyphers et al. 2015). Natural wetlands can protect upland communities by attenuating waves, capturing and accreting sediment, reducing erosion, and mitigating storm surge (Spalding et al. 2014). The degradation of coastal wetlands from human activities has slowed in the USA (Dahl 2011), but wetlands are still threatened by sea-level rise and erosion (Kirwan and Megonigal 2013; Blankespoor et al. 2014). Permanent armored structures alter connectivity through habitat fragmentation and block the landward migration of natural shoreline habitats in response to sea-level rise ("coastal squeeze"), leading to further loss of coastal habitat (Doody 2004; Bozek and Burdick 2005; Pontee 2013). Thus, there continues to be a need to conserve and restore coastal wetlands (Zedler 2004; Wigand and Roman 2012; Thorne et al. 2015), while identifying and disentangling the major drivers of wetland loss.

In this issue, Gehman et al. (2017) (*Effects of small-scale armoring and residential development on the salt marsh-upland ecotone*) compared the biota and physical and environmental characteristics of 60 tidal wetlands adjacent to bulkheads, residential backyards with no armoring, or intact forests in a cross-disciplinary study along the US Georgia coast. Elevation was lower in wetlands adjacent to bulkheads, and these sites had features typical of lower elevation wetland habitats. Wetlands adjacent to unarmored residential sites had higher soil water content and lower porewater salinity than armored or forested sites, indicating increased freshwater input to the sites. Along the wetland-upland ecotone, deposition of wrack was negatively related to elevation at armored sites but positively related at unarmored residential and forested sites. Densities of the high marsh crab *Armases cinereum* (a species that moved readily across the wetland-upland ecotone at forested sites) were lower at armored and unarmored residential sites. These results suggest that minimizing armoring while retaining some forest vegetation at the wetland-upland interface could reduce these impacts.

Human-altered ecosystems can be more susceptible to invasion by non-native species than natural systems (Hobbs 2000). A particularly aggressive invasive species in disturbed coastal systems is the European haplotype of *Phragmites* (Lelong et al. 2007). This species invades human-dominated landscapes, such as highway corridors (Jodoin et al. 2008; Leblanc et al. 2010), levees along impoundments (Bart and Hartman 2000), developed shorelines (Silliman and Bertness 2004), and constructed wetlands (Havens et al. 2003). In addition, *Phragmites* has invaded natural wetlands along the US east coast (Chambers et al. 1999) and in the continental US (Kulmatiski et al. 2010; Larson et al. 2011; Bourgeau-Chavez et al. 2013). Invasion by *Phragmites* can negatively impact wetland structure and function by altering dynamics in fish (Jones et al. 2014; Jones and Able 2015) and bird (Prosser et al. 2017, this issue) communities. However,

the consequences of invasion may not all be negative because *Phragmites* patches are not biological deserts, especially for animals (Kiviat 2013; Dibble and Meyerson 2014). The high growth rate of *Phragmites* may also contribute to sequestration of carbon and nitrogen (Kiviat 2013) and enable wetlands to keep pace with sea-level rise and minimize resulting wetland loss (Rooth and Stevenson 2000), which may be especially important in regions where the rate of sea-level rise is high (Williams 2013).

Many studies have explored the underlying causes for the rapid expansion of *Phragmites*. Earlier research focused on rhizome dispersal and establishment (e.g., Philipp and Field 2005; Bart et al. 2006), but it is now clear the primary mode of spread is the production, dispersal, and establishment of plants from seeds (e.g., Wilcox 2012). The mechanisms behind the rapid expansion of *Phragmites* have been particularly well studied in Chesapeake Bay brackish tidal wetlands, and three key factors appear to promote local expansion. First, natural and human-related disturbances create physical spaces where seedlings can become established. Second, multiple genotypes must be present because *Phragmites* can only produce seeds by cross-pollination between different genotypes (Kettenring et al. 2011). Third, elevated nitrogen levels promote expansion because seedlings grow faster, more ramets produce inflorescences, and inflorescences have more flowers, resulting in higher seed production (Kettenring et al. 2015).

Watershed land use has also been important to the expansion of *Phragmites* through its relationship to physical disturbance and elevated nitrogen. The abundance of *Phragmites* in brackish wetlands was positively related to the amounts of developed and agricultural land in watersheds that drained into the wetlands, and the closer those land use types were to the subestuary, the higher was the abundance of *Phragmites* (King et al. 2007). Sciance et al. (2016) found that agricultural land use and shoreline armoring were significant predictors of *Phragmites* occurrence; and Silliman and Bertness (2004) linked the abundance of *Phragmites* to the development of adjacent upland habitats and nitrogen loading to an estuary.

Management strategies to control *Phragmites* have been proposed (Hazelton et al. 2014; Kowalski et al. 2015), but there is little information on recovery of native vegetation following *Phragmites* removal. In this issue, Hazelton et al. (2017) (*Spatial and temporal variation in brackish wetland seedbanks: implications for wetland restoration following* Phragmites *control*) investigated whether management efforts to remove *Phragmites* may act as a disturbance, potentially fostering reinvasion. They surveyed the soil seedbank of three vegetation cover types in plots from which *Phragmites* had been removed by 2 years of herbicide treatment, plots in which *Phragmites* was left intact, and plots with native, reference vegetation in five Chesapeake Bay subestuaries. Surprisingly, the seedbank composition was not influenced

by vegetation cover type. Instead, seedbanks in plots within a subestuary resembled each other more than did treatments across subestuaries, indicating that seedbanks are fairly homogenous within a subestuary. While these diverse seedbanks could lead to invasion-resilient wetland communities, the prevalence of undesirable species in the seedbank may speed up their re-establishment following invasive plant removal if passive revegetation is the primary means of restoration.

## Stressors of Submerged Aquatic Vegetation (SAV)

SAV is a foundational component of coastal systems, and it delivers many ecosystem services. SAV provides structural habitat and refuge from predators for invertebrates and small fish (Hemminga and Duarte 2000; Gillanders 2006) as well as providing foraging habitat for waterbirds (Erwin 1996; Valentine and Heck 1999). It also stabilizes benthic substrates and moderates shoreline erosion (Madsen et al. 2001; de Boer 2007), and it improves water quality and clarity by removing nutrients (Wigand et al. 2001; Moore 2004), reducing $CO_2$-driven acidification (Unsworth et al. 2012), and reducing resuspension of sediments (Gruber and Kemp 2010). However, SAV is imperiled worldwide by multiple stressors, including sediment and nutrient runoff, physical disturbance, and climate change (Duarte 2005; Orth et al. 2006; Waycott et al. 2009; Short et al. 2011; Lefcheck et al. 2017; Orth et al. 2017). In many estuaries, the spatial extent of SAV is far below historic levels (Orth and Moore 1983; Waycott et al. 2009; Orth et al. 2010; Short et al. 2014), and increasing SAV coverage is frequently a central goal of restoration efforts (Batiuk et al. 2000; Chesapeake Bay Executive Council 2003; van Katwijk et al. 2009; Chesapeake Bay Program 2014; Orth et al. 2017).

The linkages between water clarity and SAV are well understood (Boesch et al. 2001; Kemp et al. 2004; Kemp et al. 2005; Waycott et al. 2005; Greening and Janicki 2006; Patrick and Weller 2015; Orth et al. 2017). However, the complex relationships between SAV abundance and land use distributions have only recently been quantified (Li et al. 2007; Patrick et al. 2014, 2016). SAV abundance is generally higher in estuaries with watersheds that are forested than those dominated by development and agriculture (Li et al. 2007; Patrick et al. 2014), but effects may vary with weather patterns and SAV community type. For example, agricultural impacts on SAV may be stronger in wet years when higher runoff from agricultural lands delivers more nutrients and sediments to subestuaries (Li et al. 2007). In Chesapeake Bay, the negative effects of cropland or developed land are generally stronger in polyhaline SAV communities, possibly because polyhaline SAV species are more sensitive than oligohaline species to stressors that reduce water clarity (Patrick et al. 2014; Patrick and Weller 2015).

Shoreline armoring also has negative impacts on SAV (Gabriel and Bodensteiner 2012; Findlay et al. 2014; Patrick et al. 2014; Patrick and Weller 2015). Armoring is associated with erosive habitat loss, deepening of the nearshore zone, and increased suspended sediments (Pilkey and Wright 1988; Miles et al. 2001; Findlay et al. 2014) that reduce growth and survival of SAV (Moore et al. 1997; Moore and Jarvis 2008). However, the effects of armoring also vary with land use, type of shoreline armoring, and SAV community type (Patrick et al. 2014; Patrick and Weller 2015). Patrick et al. (2014) reported that the SAV abundance throughout a subestuary was negatively related to the percentage of the riprap along the shoreline, but that this relationship could be confounded with the effects of other stressors (e.g., higher wave energy, more developed land, more boat traffic, etc.) that are correlated with the percentage of the riprap. In another study, SAV abundance was lower adjacent to armored shorelines in some settings, but the effects of armoring differed among salinity zones and dominant land use in local watersheds (Patrick et al. 2016).

Two papers in this special issue explored the complex relationships between land use, shoreline armoring, and SAV abundance. Building on previous work (Patrick et al. 2014, 2016, Patrick and Weller 2015), Patrick et al. (2017) (*Land use and salinity drive changes in SAV abundance and community composition*) analyzed long-term SAV data to better understand how watershed land use, shoreline alteration, salinity, and SAV community composition affect trends and inter-annual variation in SAV abundance. An ordination identified five SAV community types throughout Chesapeake Bay subestuaries. Some subestuaries maintained the same community through time, but in others, the community type changed across years. Community type influenced the rate of SAV recovery in subestuaries more strongly than the other stressors, but the amounts of urban or agricultural land in the local watershed did significantly reduce the rate of recovery. The strength of the land use effect differed among community types and was strongest in the mixed freshwater community.

Landry and Golden (2017) (In-situ *effects of shoreline type and watershed land use on submerged aquatic vegetation habitat quality in the Chesapeake and Mid-Atlantic coastal bays*) sought to determine if shoreline armoring degrades SAV habitat quality at the local scale. *In situ* comparisons of SAV beds adjacent to natural and armored shorelines in 24 subestuaries throughout the Chesapeake and Maryland's Coastal Bays indicated that species diversity, density, and bed width were significantly reduced by the presence of riprap. A post-hoc analysis also confirmed that human-dominated land use in the surrounding watershed negatively influenced SAV habitat. These results suggest that the expansion of shoreline armoring and human-dominated land use will threaten SAV in this region.

## Effects of Shoreline Armoring and Land Use on Benthic Fauna and Fish

A wide variety of benthic invertebrates, fish, and crabs utilize shallow-water, nearshore habitats for foraging, nursery grounds, or refuge from predation (Ruiz et al. 1993; Beck et al. 2001; Seitz et al. 2014). These organisms are key components of estuarine food webs and also include many commercially important species (King et al. 2005), so the loss or degradation of coastal habitats in association with human land use activities and shoreline armoring may have profound impacts on estuarine faunal communities.

Faunal communities in shallow-water habitats can be severely affected by shoreline modifications. For example, diversity and biomass of benthic fauna are often lower along armored compared to natural shorelines (Tourtellotte and Dauer 1983; Seitz et al. 2006; Bilkovic and Roggero 2008; Morley et al. 2012; Balouskus and Targett 2016; Gittman et al. 2016b). Armored shorelines can impact fauna by reducing vegetated habitats and deepening adjacent waters (Miles et al. 2001; Bozek and Burdick 2005; Dugan et al. 2008; Walker et al. 2011) or introducing toxins in chemically treated construction materials (Weis et al. 1998). Epibenthic predators can be influenced by bottom-up control, so they may be reduced in habitats with depauperate benthic infaunal communities (Bilkovic et al. 2006; Seitz et al. 2006; Davis et al. 2008). However, the effects on larger nekton may not be evident until three or more years after shoreline habitat alteration (La Peyre et al. 2014).

At larger scales, the degradation of water quality associated with upland development and agricultural use of watersheds can lead to less diverse habitats, reductions in nearshore faunal abundance, biomass and diversity, and systems with less-complex food webs (Holland et al. 2004; Kemp et al. 2005; King et al. 2005). Increased runoff of nutrients, pollutants, and contaminants from upland development (Jordan et al. 1997; Gregg et al. 2015) can reduce the abundance of pollution-sensitive species (Weisberg et al. 1997; King et al. 2005). Agricultural nutrient runoff also can lead to hypoxic events in warm, shallow waters (Tyler et al. 2009), which can negatively affect benthic invertebrates and nekton (Rabalais and Turner 2001; Eby and Crowder 2004; Diaz and Rosenberg 2008; Breitburg et al. 2009). In some areas where as little as 12% of the watershed was developed, the benthic community was negatively impacted (Dauer et al. 2000; Bilkovic et al. 2006). These larger landscape-level effects may either mask (Lawless and Seitz 2014; Lovall et al. 2017) or exacerbate (Seitz and Lawless 2008; Seitz et al. 2017, this issue) the effects of smaller-scale habitat alterations (e.g., shoreline armoring) on benthic infauna, fish, and crab abundance.

Six papers in this special issue investigated the effects of land use or shoreline armoring on macrofaunal communities. Seitz et al. (2017) (*Human influence at the coast: Upland and shoreline stressors affect coastal macrofauna and are mediated by salinity*) examined how subestuary land use and shoreline development affected density, biomass, and diversity of benthic infaunal communities in 14 subestuaries of Chesapeake Bay. Benthic biomass was significantly lower in subestuaries with developed compared to forested upland use. Infaunal biomass declined exponentially with the proportion of nearshore developed land because large, pollution-sensitive species declined, whereas the density of small, pollution-tolerant species increased. For large macrofauna (> 3 mm), density did not differ significantly among natural wetland, beach, and riprap habitats, but tended to be lower adjacent to bulkhead shorelines, particularly in low-salinity (< 15 psu) subestuaries. Benthic diversity tended to be highest adjacent to natural wetlands compared to other habitats in low-salinity subestuaries. Upland and shoreline development were both important in driving patterns in benthic community structure in multi-model comparisons. These results suggest that minimizing subestuary upland development and shoreline armoring may be essential for maintaining productive benthic communities within the estuarine food web.

Crum et al. (2017) (*Growth and movements of mummichogs* (Fundulus heteroclitus) *along armored and vegetated estuarine shorelines*) assessed the impact of shoreline alteration on an estuarine fish by comparing growth and movement along stretches of shorelines dominated by *Spartina alterniflora*, *Phragmites*, riprap, and bulkhead in a tributary of the Delaware Coastal Bays. Fish were individually tagged and displayed a high degree of site fidelity overall. Growth rates were greatest along riprap, intermediate at bulkhead, and lowest along *Spartina*- and *Phragmites*-dominated shorelines. Combining these growth rates with density data from a concurrent study (Balouskus and Targett 2017, this issue) yielded the highest estimates of relative *F. heteroclitus* productivity along *Spartina* and *Phragmites* shorelines, intermediate at riprap, and lowest along bulkhead, suggesting that armoring reduces abundance sufficiently to negatively affect localized productivity of this species.

Balouskus and Targett (2017) (*Impact of armored shorelines on shore-zone fish density in a mid-Atlantic, USA, estuary: modulation by hypoxia and temperature*) demonstrate how nearshore habitat use by estuarine fish species is driven by both physical habitat structure (shoreline structure) and water characteristics such as dissolved oxygen and temperature. During times of low dissolved oxygen and increased water temperature, many fish species had higher fidelity to natural wetland habitat (*Spartina alterniflora*) compared with armored (riprap, bulkhead) and *Phragmites*-dominated shorelines. This suggests that a combination of stressors, including altered shorelines and reduced water quality, affects fish habitat usage.

Kornis et al. (2017) (*Shoreline hardening affects nekton biomass, size structure and taxonomic diversity in nearshore*

*waters, with responses mediated by functional species groups*) evaluated mobile fish and shellfish assemblages within two nearshore habitat zones adjacent to natural (native wetland or beach) and armored (bulkhead or riprap) shorelines. Within 0–3 m from shore, total biomass was greatest at bulkhead and riprap shoreline types, driven by species from the planktivore (e.g., bay anchovy) and benthivore-piscivore (e.g., white perch) functional groups, whereas small-bodied littoral-demersal species (e.g., *Fundulus* spp.) had greatest biomass at wetland habitat. In contrast, total fish biomass was comparable among all four shoreline types within 16 m from shore, suggesting the effect of shoreline armoring on fish biomass is most pronounced immediately at the land/water interface. Utilization of shoreline type was also mediated by body size across all functional groups, with individuals ≤ 60 mm most abundant at wetlands and beaches and individuals > 100 mm most abundant at armored shorelines. Taxonomic diversity was lowest at beach shoreline types within 0–3 m from shore, but greatest at wetlands and beaches within 16 m from shore. The study highlights substantial differences in fish and mobile shellfish biomass density along armored versus natural shorelines, and illustrates how assemblage composition and diversity are mediated by both shoreline type and body size.

Dugan et al. (2017) (*Generalizing ecological effects of shoreline armoring across soft sediment environments*) compared the effects of shoreline armoring structures across soft sedimentary habitats using a conceptual model that scaled the predicted ecological effects of armoring based on its purpose (e.g., to slow or stop water) and on wave energy at the structure. Using a suite of six ecological responses to shoreline structures (changes in habitat distribution, species assemblages, trophic structure, nutrient cycling, productivity, and connectivity), the model predicted that the ecological impacts of armoring will be larger for structures built to stop as opposed to slow water, and will increase with increasing energy. A literature review of the documented ecological effects of different types of shoreline structures (including living shorelines) across a range of environments (beaches, harbor shores, salt marshes, and mangroves) showed that 70% of the reported effects were significantly negative. However, the percentage of negative responses varied with the intended effect of the structure on water flow, with higher percentages observed for structures designed to stop water flow. Furthermore, the highest percentage of negative responses was reported for high-energy environments. These results suggest that the conceptual model is useful for predicting the relative impact of structures.

The use of living shorelines by fauna has only recently been investigated, and several studies have demonstrated improvements in diversity and abundance compared to traditional armored shorelines (Currin et al. 2010; Bilkovic and

Mitchell 2013; Balouskus and Targett 2016; Gittman et al. 2016a). However, further studies are needed to examine the ecological effects of living shorelines. Some hybrid armored features (e.g., marsh-sills) may provide improved habitat relative to traditional riprap, but only when they have a small footprint that minimally impacts the infauna displaced when structures are installed on the sediment surface (Bilkovic and Mitchell 2013). In this issue, Davenport et al. (2017) (*Living shorelines support nearshore benthic communities in upper and lower Chesapeake Bay*) monitored biological responses of shallow estuarine benthic communities to living shoreline construction for two case studies in Chesapeake Bay with before-after control-impact study designs. Communities of large (> 3 mm) infauna adjacent to living shorelines at both study locations increased in biomass within 3 years of installation. Community assemblages and metrics (density, biomass, and species richness) showed similar trajectories at both living shoreline and fringing wetland control shorelines toward the end of the study period, suggesting that relatively mature living shorelines and fringing wetland shorelines support similar benthic infaunal communities. These results illustrate the need for long-term studies to understand the impacts of living shorelines on benthic communities.

## Effects of Shoreline Armoring and Land Use on Waterbirds

Waterbirds, including waterfowl, shorebirds, marsh birds, seabirds, and wading birds, also respond to changes in ecological integrity associated with shoreline armoring and land use. Birds utilize a wide range of nearshore estuarine habitats including wetlands, intertidal flats, beaches, SAV, and shallow waters for foraging, breeding, nesting, roosting, migratory staging, and overwintering (Erwin 1996; Ma et al. 2010). In addition, waterbirds often occupy the highest trophic level in these ecosystems, so their abundance and diversity reflect conditions influencing lower trophic levels (Takekawa et al. 2006). Consequently, the loss or degradation of coastal habitats may have severe implications for waterbird populations (Erwin et al. 2011).

Waterbird communities can be negatively impacted by anthropogenic land use within the surrounding watershed (DeLuca et al. 2004; Shriver et al. 2004; DeLuca et al. 2008; Smith and Chow-Fraser 2010; Studds et al. 2012). Expansion of urban development and agricultural fields leads to fragmentation and loss of wetlands, which can reduce waterbird species diversity, overall abundance, and abundance of specialist species (Alsfeld et al. 2010; Celdrán and Aymerich 2010; Ludwig et al. 2010). Other stressors, such as nutrients and toxins in urban and suburban runoff, may influence waterbirds directly (Kushlan 1993) or indirectly, through reductions in prey abundance and diversity (Martínez Fernández et al. 2005; Bilkovic et al. 2006). The

spread of invasive *Phragmites* may reduce habitat quality and function for some marsh waterbird species (Benoit and Askins 1999). These stressors may interact with habitat fragmentation and loss to further reduce waterbird community integrity.

At local scales, shoreline armoring may also negatively influence waterbird communities. Artificial structures lack the structural complexity and refuge function of natural shorelines, so they support different assemblages of epibiota and fish with reduced biodiversity relative to assemblages on natural shorelines (Seitz et al. 2006; Bulleri and Chapman 2010; Sobocinski et al. 2010; Strayer et al. 2012; Heerhartz et al. 2014, 2016). Consequently, armored shorelines offer reduced waterbird foraging habitat quality, which can lead to reductions in waterbird abundance and diversity (Dugan and Hubbard 2006; Dugan et al. 2008). In addition, shoreline armoring can lead to losses of foraging (Bulleri and Chapman 2010; Dugan et al. 2011) and roosting habitats (Erwin et al. 2006; Wilson et al. 2007; Clausen and Clausen 2014) by preventing landward migration of tidal wetlands and by deepening nearshore waters due to increased wave scour and reduced sediment inputs.

In this issue, Prosser et al. (2017) (*Effects of local shoreline and subestuary watershed condition on waterbird community integrity: influences of geospatial scale and season in the Chesapeake Bay*) examined waterbird community integrity during two seasons (late summer and late fall) in relation to shoreline armoring and land use characteristics in 21 Chesapeake Bay subestuaries. They considered three scales: (1) the shoreline scale characterized by seven shoreline types: bulkhead, riprap, developed, natural wetland, *Phragmites*-dominated wetland, sandy beach and forest; (2) the 500 m landscape scale surrounding the shoreline edge; and (3) the watershed scale. They incorporated waterbird survey data into an Index of Waterbird Community Integrity (IWCI) based on sensitivity to human disturbance to characterize the communities in each subestuary and season. Multivariate analysis showed that the local shoreline scale had the strongest influence on IWCI, and percent bulkhead and percent *Phragmites* along the shoreline were the strongest predictors of IWCI. In addition, low thresholds existed for *Phragmites* (5%) and for bulkhead (8%), beyond which IWCI declined. Their results indicate that local-scale shoreline armoring and the presence of invasive *Phragmites* have a negative effect on waterbirds using subestuarine systems.

## Management Approaches

Effective management of estuarine and coastal ecosystems requires knowledge of community dynamics and ecological processes. However, estuarine science often is not effectively integrated into coastal management policies due to inherent obstacles between the scientific and management communities (e.g., institutional boundaries, governance issues, and the lag time between producing scientific results and developing management policies) (Leslie and McLeod 2007; ORAP 2013; Cvitanovic et al. 2015; Zedler 2017). In this issue, Turner and Jordan (2017) (*Integrating management needs into a Mid-Atlantic shorelines research project*) explored how the addition of management advisors to an estuarine science project influenced the direction of the science and the dissemination of results to interested stakeholders. They outlined the genesis and functions of the management advisory committee for a project dealing with the ecosystem effects of different shoreline and watershed types in the Mid-Atlantic region, and put the function of the committee in the context of previous reports that recommend the integration of science and management. This approach included (1) a well-targeted request for proposals; (2) a review process that included management input; (3) a process for selecting advisory committee members at the appropriate level in the agencies; (4) regular meetings between the advisory group and the science team through the duration of the project; and (5) active involvement of a program manager as the project liaison. Engagement of a management advisory group led to adaptations in scientific sampling and analyses to better reflect management interests, improved communication of results with managers, and formed a foundation for incorporation of the findings into regional management and initiatives.

## Summary

The articles in this special issue advance the science and management of coastal systems in several ways. First, they quantify consistent negative impacts of agricultural and developed land and of shoreline armoring on nearshore water quality, estuarine habitats, and fauna. Second, they document the need for improved management to minimize the impacts of shoreline armoring, to conserve and restore coastal habitats, and to maintain valued ecosystem services in the face of changing land use and rising sea levels. Third, they highlight the need for more research on innovative practices, like living shorelines, which seek to stabilize shorelines while preserving habitat. Finally, they demonstrate the value of engaging environmental managers in planning and executing research programs. The studies in this issue focused primarily on changes in habitat and species distribution, leaving fundamental gaps in our knowledge of how shoreline armoring affects other ecological processes like nutrient cycling, productivity, and trophic interactions. Future studies could benefit by sampling shorelines before construction is initiated (using a before-after control-impact design) and by incorporating long-term monitoring, which is key to understanding overall ecosystem effects of anthropogenic alterations of the landscape.

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