

8.02 Estuarine and Coastal Structures: Environmental Effects, A Focus on Shore and Nearshore Structures

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Abstract

Rapidly growing populations and expanding development are intensifying pressures on coastal ecosystems. Sea-level rise and other predicted effects of climate change are expected to exert even greater pressures on coastal ecosystems, exacerbating erosion, degrading habitat, and accelerating shoreline retreat. Historically, society's responses to threats from erosion and shoreline retreat have relied on armoring and other engineered coastal defenses. Despite widespread use on all types of shorelines, information about the ecological impacts of shoreline armoring is quite limited. Here we summarize existing knowledge on the effects of armoring structures on the biodiversity, productivity, structure, and function of coastal ecosystems.

8.02.1 Introduction

Rapidly growing populations and expanding urbanization and land development are intensifying pressures on coastal ecosystems worldwide (Clark, 1996). Sea-level rise and other predicted effects of climate change are expected to exert even greater pressures on these important ecosystems, exacerbating erosion, degrading habitat, and accelerating rates of the landward retreat of shorelines (Nordstrom, 2000; Slott et al., 2006).

Throughout history, estuaries and coastal embayments have been centers of human settlement and commerce, leading to the development of many very large coastal cities all over the world (Mann, 1988), a trend that continues (Yapp, 1986; Suchanek, 1994; Burke et al., 2001; Lotze et al., 2005). These populous cities require expansive infrastructure, developing shorelines and reclaiming intertidal and shallow subtidal areas to meet growing societal needs. The majority of exposed sandy coasts are already classified as eroding (Bird, 2000) and

this also applies to the shorelines of estuaries and bays (Harmsworth and Long, 1986; Allen, 2000; van der Wal and Pye, 2004). The infrastructure associated with urbanization and other human interventions in coastal processes (including human-induced land subsidence and reclamation, offshore and channel dredging, decreased sediment supply from rivers, and destruction of seagrass meadows, marshes, beaches, and coastal sand dunes), together with poor coastal defense policies, has, directly or indirectly turned coastal erosion into a problem of mounting intensity and concern (French, 1997; Kennish, 2002; EC, 2004; Defeo et al., 2009).

Global climate change and sea-level rise are beginning to pose severe threats to beaches, salt marshes, coastal wetlands, and river deltas (Adam, 2002; Morris et al., 2002; Schlacher et al., 2007). On undeveloped coasts, losses from sea-level rise and increased erosion could be compensated for by the habitat regressing or retreating landward. However, in populated areas where coastal boundaries are developed and often defended by



Figure 1 The developed shoreline at Surfers Paradise, Australia, provides an extreme example of coastal squeeze.

man-made barriers, coasts are caught in a coastal squeeze between rising seas and expanding development (Doody, 2004; Schlacher et al., 2007) (Figure 1). Projections of loss relative to possible future changes in sea level, recession of coastlines, and the coastal squeeze are available for some coastal areas. Coastal squeeze threatens to eliminate sandy beaches from large stretches of shoreline over the next 50–100 years (Schlacher et al., 2007; Defeo et al., 2009). Projected sea-level rise could cause the loss of up to half of the existing European coastal wetlands (EC, 2004), with some of the largest losses expected to occur around the Mediterranean and Baltic seas (Nicholls et al., 1999). When combined with other losses directly or indirectly related to human action, up to 70% of the world's remaining coastal wetlands could be lost within the next 100 years (Nicholls et al., 1999), although there is considerable uncertainty. As an example, a loss of freshwater and brackish habitats of around 4000 ha as a consequence of the combined effects of sea-level rise and a temperature increase of 3–4 °C has been predicted for the United Kingdom (Lee, 2001). As well, a loss of 80–100 km² of intertidal flats is predicted to occur between 1993 and 2013 (UK Biodiversity Group, 1999), particularly in southern and southeast regions.

In response to coastal erosion and related hazards, coastlines have been actively defended with engineered structures ever since mankind settled in organized societies in the coastal zone (Charlier et al., 2005). Thus, these armoring structures are not a recent phenomenon, although their extent and size have increased dramatically in recent decades. Despite growing

evidence and awareness of the impacts of the substantial environmental impacts of coastal cities and infrastructure, major changes to coastlines continue unabated (Mann, 1988). However, detailed information on either the extent of changes or the specific ecological effects of such changes is seldom available (Bulleri, 2006; Airolidi and Beck, 2007)

8.02.2 History and Use of Shore Structures

Society's responses to threats to infrastructure and development from coastal erosion and shoreline retreat have historically relied on armoring and other engineered forms of coastal defense built to slow down or halt loss and migration of the shoreline (Nordstrom, 2000; Rippon, 2000; Charlier et al., 2005; Griggs, 2005a, 2005b). Early forms of armoring included breakwaters and other structures built to stabilize harbors along Mediterranean coasts by 2 BC, and large coastal defense projects initiated in China by 25 BC (Charlier et al., 2005). As early as 175 BC, earthen mounds or dams were constructed along the coast of the Netherlands in attempts to protect low-lying coastal land and towns (Rippon, 2000). This approach to managing shorelines was well established by the 1200s in Northern Europe (Charlier et al., 2005), both in estuaries where it was extensively used for land reclamation in coastal marshes and on exposed coasts. By the Middle Ages, seawalls were in common use in Europe, although they were probably used much earlier in the Middle and Far East. Groynes were similarly used in Europe from at least the 1850s, but again probably appeared earlier in other regions.

Shorelines are being increasingly hardened worldwide, although this trend is most noticeable on developed and urbanized coastlines (Nordstrom, 2000; Airolidi et al., 2005a). Shore armoring is used both in sheltered estuaries and bays and in open-coast settings where erosion due to wave action may be more of a problem. Nevertheless, estuarine shores are particularly affected by urban infrastructure and armoring. Recognition of this phenomenon is not new. In fact, in 1844, one of the reasons given by William Cullen Bryant for the establishment of Central Park (New York) was for “one place where tides may be allowed to flow pure, and the ancient brim of rocks which borders the waters left in its original ...,” a response to the proliferation of docks along the shoreline (Laurie, 1979).

Coastal defense and armoring structures are deployed on all types of open and sheltered coasts in a wide range of tidal and wave conditions, as well as in onshore and offshore locations. The majority, especially those constructed for protection against erosion, are constructed upon coastal landforms dominated by soft sediments, including beaches, dunes, friable coastal bluffs, estuarine and tidal creek channels, mudflats, harbors, and inlets (Nordstrom, 2000). Armoring is most often applied in attempts to reduce erosion and flooding threats to coastal developments, infrastructure, and high-value coastlines (Charlier et al., 2005), but is widely used at inlet and harbor mouths to maintain channels for shipping and navigation. These structures are also built to stabilize and retain beaches and reclaimed land, and to increase the amenity value of the coast (e.g., tourism, beach use, and surfing) (Walsh et al., 2004; Airolidi et al., 2005a). Coastal armoring, not constructed specifically to counteract erosion, that results from expansion of urban infrastructure (e.g., piers, docks, wharves, promenades and marinas), can be built over hard or

soft substrata however is generally more extensive in sheltered areas.

Major armoring efforts by coastal communities have often followed a devastating storm or flood event. For example, the construction of one of the most extensive coastal defense structures ever built, the system of dykes/dams known as the Delta Works in the South of Holland (Province of Zeeland), was initiated following the North Sea Flood tidal surge disaster of 1953 that breached existing coastal defenses and claimed more than 2000 lives in the region (Kabat et al., 2009). The 4.8-km-long seawall on the ocean shore of Galveston, Texas in the United States, was erected following a major hurricane and 4.6-m storm surge in 1900 that killed more than 6000 people on the barrier island (Hansen, 2007). The seawalls that now surround the capital island of Malé in the Maldives were built following tidal surges that flooded the capital in 1987, causing millions of dollars in damage (Harangozo, 1992).

Coastal defense structures, such as these, may impart a misplaced sense of safety from storm surges, floods, and waves to coastal cities and landowners, even leading to expanded shoreline development in some regions. The dynamics of the coastline mean that continued maintenance and renovation of these structures are required and there may be major effects of the structures on adjacent or downcoast shores that need to be addressed. Failure rates of coastal armoring from scour, or undermining, outflanking, overtopping, and battering by storm waves, are relatively high, particularly for low-budget efforts (Griggs, 1999). Even large well-engineered structures can experience overtopping by waves and catastrophic failure (Griggs, 1999) with risks not only to infrastructure but also to human safety. There is thus always ongoing need to monitor, repair, and maintain such structures, which is costly. It was estimated in 1991 that the United Kingdom was spending more than US\$1.5 million per km for coastal erosion defenses that were expected to last only 50 years (Jones, 1994). Much of the responsibility for maintaining or building seawalls, groynes, etc. belongs to government authorities, for which repairs and replacement of seawalls can form a major part of their budget (M.G. Chapman, personal observation). These costs are ultimately, one way or another, picked up by the general public.

8.02.3 Types of Structures

Design and engineering of coastal armoring structures, and the materials used to build them, vary widely, as do their costs, efficacy, and life span (Jones, 1994; Griggs, 1998). In general, however, coastal armoring entails the placement of resistant artificial structures, such as groynes, jetties, dykes, seawalls, and other engineered designs, which may be constructed of stone, concrete, wood, steel, or geotextiles. We briefly describe armoring structures in a few broad categories in order to explore some general themes with regard to their effects on coastal environments.

8.02.3.1 Shoreline Structures

Alongshore structures, including seawalls, revetments, and bulkheads, are built parallel to the shoreline. Usually, these are constructed to protect coastal development and infrastructure from erosion or wave attack after loss or movement of the

original shoreline (Weigel, 2002a); in more sheltered settings, they are used to protect the edges of reclaimed land. Seawalls and revetments are usually built as barriers to wave action on exposed shorelines, while in ports and harbors of estuaries they can also provide access to land for loading and unloading of ships. Bulkheads are generally built to function more as retaining walls in these sheltered waters.

Seawalls are mostly vertical or steeply curved solid structures usually made of timber, concrete, or tightly interlocked stone, although a wide variety of materials have been used (Figure 2). Their foundations directly cover and reduce soft-sediment intertidal habitat, but they may create both intertidal and subtidal hard substrata because they are usually built from the seafloor to above the high water level. Bulkheads are also vertical structures, made of wood or other hard materials that resemble retaining walls, but they are often initially placed above mean high water and landward of the beach or backfilled (Figure 3). However, along sheltered shores, such as estuaries and tidal channels, bulkheads can also be placed lower on



Figure 2 An intertidal concrete seawall located on a beach along the open coast of Santa Barbara County in southern California, USA.



Figure 3 An example of a bulkhead built to protect coastal parkland at Redland Point, Queensland, Australia.



Figure 4 An intertidal bulkhead constructed of metal sheet pilings located on Puget Sound, Washington, USA. This bulkhead was built to retain contaminated sediments and reduce exchange with the sound.



Figure 6 Beach-filling activities between rock groynes on the open coast of Galveston, Texas, USA in January 2009 following severe erosion and damage from Hurricane Ike.



Figure 5 An example of a rock revetment located on an open-coast beach of Santa Barbara County in southern California, USA.

the shore to act as a primary coastal defense (Figure 4) (Nordstrom, 2000).

Revetments, by contrast, are mostly made of large boulders (riprap) or articulated concrete blocks or tetrapods, which are either placed in a distinct structural design or simply piled up to a sufficient height (Figure 5). They may be built to similar heights as seawalls, but they have more gradual slopes and much larger structural footprints. For example, a 6-m-high revetment with a slope of 2:1 will cover 12 m of beach habitat (Griggs, 2005b).

Structures that are placed perpendicular to the shoreline (shore-normal orientation) include groynes and jetties or breakwaters. Groynes are placed on beaches either singly or in a series to create a 'groyne field' (Figure 6). Their primary purpose is to maintain the width of an upcoast beach or to control the amount of sand moved alongshore by the littoral drift (Dong, 2004). Groynes are increasingly used to maintain imported sediment for beach filling or nourishment programs in response to coastal erosion (Dong, 2004; Figure 6). Jetties or breakwaters that extend out from the shoreline at inlets or harbor mouths are used to control the flow of water and



Figure 7 A complex of coastal defense structures associated with a marina in Spain.

sediments to maintain the channels for tidal flushing and/or navigation. Jetties are also used to decrease the migration of tidal channels by reducing longshore currents and sediment transport. They are also commonly used to create access for boats, especially where the water is too shallow to allow boats access to the shore. In marinas, jetties are combined with piers and often seawalls and floating pontoons, to create large areas of built infrastructure in shallow waters immediately offshore (Figure 7). Marinas are most common in sheltered waters, such as estuaries, but on more exposed coasts are usually protected by a large offshore seawall or groyne to provide shelter for the boats when moored. Marinas can contain several different types of artificial structures that can be very extensive, covering hectares and providing berths for hundreds of yachts and small vessels (Figure 7).

Developers have responded to the increased demands for waterfront properties by creating 'canal estates' in sheltered waters (Figure 8). These are completely artificial water bodies, composed of man-made branches of land supporting houses, separated by thin channels of water. Canal estates have a major ecological footprint, because they convert large areas of natural habitat into man-made landforms and narrow channels (Long et al., 1996), and these newly created artificial habitats do not support natural populations of biota (Morton, 1992).

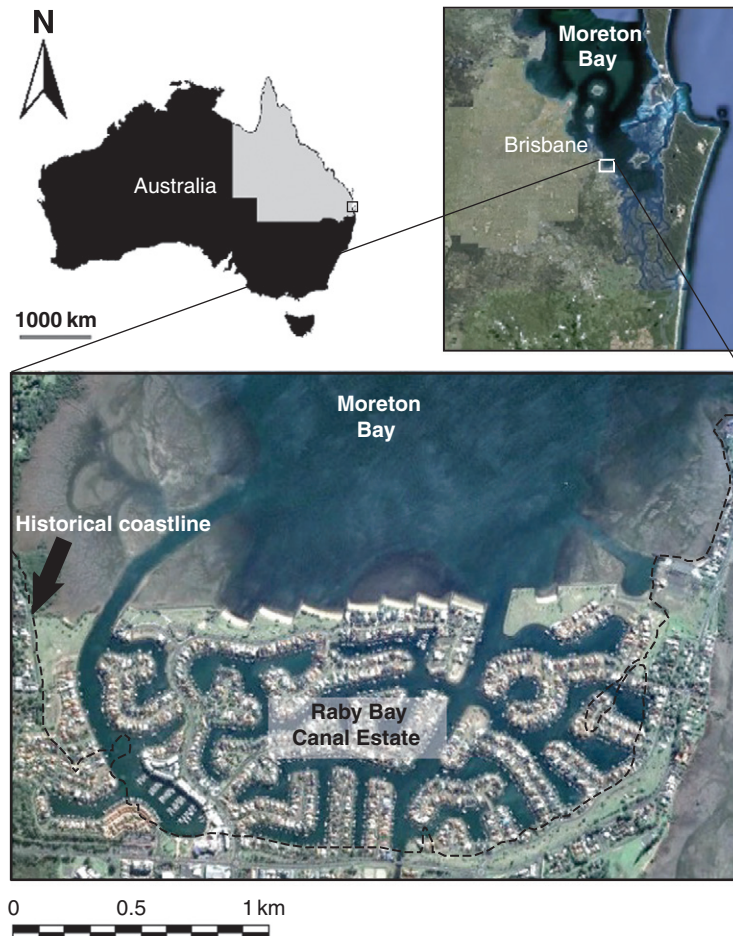


Figure 8 An example of the development of canal estates near Brisbane, Australia.

8.02.3.2 Offshore or Detached Structures

Offshore structures, including emergent and low-crested structures or detached breakwaters, are generally placed parallel to the shore in deeper water at a certain distance from the shoreline. They are more common in more exposed settings than in estuaries. Their main purpose is to reduce the rate of shoreline change or erosion by decreasing the wave energy reaching the shore through dissipation, refraction, or reflection of incoming waves (Nordstrom, 2000). The conditions of lower wave energy created enhance the deposition of sediments in the lee of the structure creating beaches that may grow seaward and, in some cases, attach to the detached structure. In some regions, such as along the open Italian coast, detached breakwaters are used in combination with beach filling to create sheltered beaches for recreational use (Figure 9). Detached breakwaters can be combined with or attached to a variety of other coastal defense structures, such as groynes and jetties. For this chapter, we are covering only offshore structures used for coastal defense. The wide variety of other offshore structures including marine energy installations (traditional gas oil, or renewable) (Page et al., 2006, 2008; Terlizzi et al., 2008; Inger et al., 2009), artificial reefs, fish aggregating devices, and other restoration structures (Baine, 2001; Perkol-Finkel et al., 2006; Miller et al., 2009) are outside the scope of this chapter.

8.02.3.3 Scope of Coastal Armoring

Infrastructure that either is built over or replaces natural habitats in order to support growing human populations on the coast, together with coastal protection and defence measures (e.g. breakwaters, groynes, seawalls, jetties, dykes, or other armoured structures) proliferated in the second half of the 20th century. This led to severe hardening of coastlines and changes in sediment dynamics in many coastal settings (Airoldi et al., 2005a). It is expected that armoring will further increase as a result of burgeoning coastal populations, expansion of coastal cities, and greater threats from climate change, storm surges, and sea-level rise. Despite the increasing prevalence of hardened and armored shorelines, particularly on urban and developed coasts across the globe (Nordstrom, 2000), there has been very little research into their environmental effects, certainly relative to the amount of research conducted on the effects of urban development on terrestrial ecosystems (Chapman and Underwood, 2009). Shoreline armoring is currently built with little or no information about the ecological impacts on coastal ecosystems, or how these artificial structures may affect biodiversity, productivity, and the provision of ecosystem functions. Here, we summarize information on the known extent of these structures for regions where sufficient data are available in Europe, North America, Australia, and Asia. Although our summary



Figure 9 Aerial view of the urban structures along the coasts of the Adriatic sea in northern Italy. Photo by Benelli, reproduced from Airolidi, L., Beck, M.W., 2007. Loss, status and trends for coastal marine habitats of Europe. *Oceanography and Marine Biology: an Annual Review* 45, 345–405.

clearly represents a snapshot in time and is likely to be an underestimate of the extent of coastal armoring in the world, it provides a reasonable basis to evaluate the scope of putative environmental effects on coastal ecosystems associated with shoreline armoring.

In Europe, >15 000 km of the coastline is actively retreating, despite coastal protection works along 2900 km (EC, 2004). Another 4700 km of coast is artificially stabilized (EC, 2004). A recent review of the status of European coastlines (Airolidi and Beck, 2007 and references therein) has shown that 22 000 km² of the European coastal zone is now covered in concrete or asphalt. Urbanization covers over 50% of the land in coastal areas in several European countries (Duarte, 2002) and, in some regions, the growth of cities, ports, tourism, and industries has led to development of over 90% of the coastline (Jeftic et al., 1990; Meinesz et al., 1991; Cencini, 1998). For example, by 1996, 42.6% of the entire Italian coast had been subjected to intensive development (completely occupied by built-up centers and infrastructure), 13% had extensive development (free zones occupied only by extensive building and infrastructure) and only 29% was free of buildings and infrastructures (reviewed in Airolidi and Beck, 2007). More than 50% of Mediterranean coastlines

are dominated by concrete structures (>1500 km), of which about 1250 km have been developed for harbors and ports (EEA, 1999). This is particularly striking in the North Adriatic Sea, where >190 km of artificial structures, mainly groynes, breakwaters, seawalls, and jetties (Figure 9), make up more than 60% of the coastline along 300 km of naturally low sedimentary shoreline (Bacchiocchi and Airolidi, 2003). Similarly, in Belgium and in the Wadden Sea region, there are no natural rocky shores and all intertidal hard substrata are created by man-made structures (Johannesson and Warmoes, 1990; Reise, 2005). Coastal zone urbanization is projected to further increase by 10–20% in the near future for most Mediterranean countries.

Despite a much briefer history of urban development, many coasts of the United States are also extensively armored. Armoring covers more than 50% of the coastline in a number of estuaries and bays, including some subwatersheds of Chesapeake Bay in Virginia, Maryland, Barnegat Bay in New Jersey, and San Diego Bay in California (Living Shoreline Summit Steering Committee, 2006). Along the Atlantic coast, ~17% of the coastline of New Jersey has been altered by the addition of bulkheads, revetments, or other coastal defense structures (Lathrop and Love, 2007). In Florida, a 1990 analysis estimated that ~21% of the 759-km coastline was armored with values of 45–50% along developed shores (Florida DEP, 1990). Trends are similar for the Pacific coastline in California, where ~12% of the 1763-km-long coast has been armored during the last century (Griggs, 1998) and the geographic extent of armoring on the coast increased by 400% between 1971 and 1992 (Griggs, 1998). In densely populated southern California, armoring covers 30% of the coastline overall (112 km of 371 km of coast), but 70% or more of the coasts of the cities of Long Beach, Seal Beach, San Clemente, and Oceanside is armored. In Oregon, where the coastal population is smaller, only 6% (35 km of 582 km) of the coastline is estimated to be armored (Surfrider, 2010). However, further north, on the sheltered shores of the Puget Sound in Washington State, 1136 km of armoring covers 30% of the 3788-km-long coastline (T. Quinn, unpublished).

These patterns are also evident in the Western Pacific. In Japan, 15 900 km of the 34 500-km coastline was estimated to be vulnerable to erosion and 27% (9400 km) had been hardened with some type of artificial structure (armoring, breakwaters, and dykes) by 1992 (Koike, 1993). Most of the population of Australia is concentrated into a few coastal cities and this has resulted in significant modifications of shorelines in urban areas (Chapman, 2003). For example, more than 50% of the shores of Sydney Harbour have been altered with either coastal infrastructure or armoring (Chapman, 2003). In parts of Australia, canal estates built in estuaries increase the area of rocky shore habitat within soft-sediment environments, modifying the coastline substantially. For example, the construction of Raby Bay in Southeast Queensland added >19 km of concrete and rock revetment walls to the existing coastline (Figure 8; data courtesy of Redland City Council, 2010). However, even largely rural coastlines can be affected, as exemplified by Fiji where localized shoreline-hardening efforts have increased dramatically since 1960 in response to erosion (Mimura and Nunn, 1998).

8.02.4 Current State of Knowledge on Environmental Effects

8.02.4.1 Alteration of Coastal Processes

Starting from first principles, any engineered structure placed in a coastal setting will alter hydrodynamics and modify the flow of water, wave regime, sediment dynamics, grain size, and depositional processes (Fletcher et al., 1997; Miles et al., 2001; Runyan and Griggs, 2003; Martin et al., 2005). For soft-sediment habitats, the loss of original habitat that is covered by the footprint of man-made coastal structures is a primary impact, along with the altered coastal hydrodynamic processes in the remaining and adjacent habitats. The effects of these physical changes on subtidal and intertidal benthic communities result in ecological changes on both open and sheltered coasts.

On open coasts, groynes, seawalls, revetments, jetties, geotextile tubes, and other engineered structures alter the wave regime and modify processes that deposit and retain mobile sediments on exposed sandy beaches (e.g., Miles et al., 2001). For alongshore structures (seawalls and revetments) placed on beaches, the hardened faces reflect wave energy and constrain natural landward migration of the shoreline, generally leading to loss of beach area and width and flanking erosion of adjacent shorelines (e.g., Hall and Pilkey, 1991; Griggs, 2005a, 2005b). Shore-normal and offshore structures, such as jetties, groynes, and breakwaters, can affect erosion and accretion of adjacent shorelines, as well as sediment transport and deposition (French, 1997; Nordstrom, 2000).

The effects of alongshore coastal armoring on the physical features of open-coast beaches are well described and documented (see reviews by Kraus and McDougal, 1996; Nordstrom, 2000; Weigel, 2002a, 2002b, 2002c; Griggs, 2005b). Beach widths are reduced seaward of shore-parallel structures, such as seawalls and revetments, initially in response to placement loss, followed by the ongoing effects of coastal processes, such as passive and active erosion (Figure 10). Placement loss, the reduction of beach area resulting from the footprint of the armoring structure, and passive erosion, in which shoreline retreat is inhibited and the beach in front of

structure drowns as adjacent shoreline migrates landward, are widely recognized effects of seawalls and revetments (Figure 10) (Hall and Pilkey, 1991; Fletcher et al., 1997; Griggs, 2005b). The importance of active erosion of the beach caused by the seawall itself is less broadly accepted (Kraus and McDougal, 1996; Griggs, 2005b). Impacts of active erosion include scour of the beach in front of the structure, as well as the effects of flanking erosion associated with stronger physical processes, such as increased wave reflection and the narrowing of the surf zone during storms (e.g., Hall and Pilkey, 1991; Griggs, 1998, 2005a, 2005b; Miles et al., 2001). These effects appear to be related to the hardened faces of seawalls which reflect rather than dissipate wave energy, combined with the constraints of armoring on natural retreat of the shoreline. Importantly, these effects scale with the degree of interaction of the structure with waves and tides. Generally, the lower a structure is located on the beach profile, the greater the physical impacts associated with it (Weigel, 2002a, 2002b, 2002c).

In coastal marshes and estuaries, placement loss and the effects of seawalls and bulkheads on the coastal processes described above, can also cause significant habitat loss, erosion, and shoreline change. Seawalls and bulkheads can alter tidal currents leading to the permanent removal of sediment from the littoral transport system or cell; this results in sediment starvation and downdrift erosion of unarmored shores, as well as altered water exchange. The reflection of nonbreaking waves from the face of seawalls or bulkheads leads to the evolution of oversteepened beach faces (NRC, 2007). As the armored shoreline erodes, the intertidal zone is reduced or eliminated with loss of sheltered beaches, oyster reefs, mudflats, and vegetated marshes (Harmsworth and Long, 1986; Douglass and Pickel, 1999).

Abrupt discontinuities in shoreline orientation and truncation of downcoast beach profiles can be produced by groynes and jetties (Nordstrom, 2000). Shoreline erosion can be greatly accelerated downcoast of shore-normal structures, such as groynes and jetties, with long-term erosion rates of 6–11 m yr⁻¹ reported (Nordstrom, 2000). To lessen this problem, permeable groynes that allow some littoral transport of sediments to continue, have been deployed along some coastlines, particularly along the Polish and German coasts (Nordstrom, 2000). Reducing the height of groynes to offset impacts sediment transport and downcoast erosion has also been tried in some areas. Groynes and jetties also change wave regimes and surf zone circulation, creating new rip currents and altering the benthic topography of the seafloor, with features such as deep holes and depositional lobes forming adjacent to the structures (Sherman et al., 1990; Pattiaratchi et al., 2009). Regular or, in some regions with strong littoral currents, continuous dredging or bypassing is often needed to move trapped sediments across an inlet or harbor mouth that has been stabilized with jetties (e.g., Patsch and Griggs, 2008).

The effects of offshore structures, including emergent and low-crested structures and detached breakwaters, which are generally placed parallel to the shore in deeper water, can also cause significant shoreline change. Sheltered beaches or salients can rapidly develop inshore of the structures on open coasts (Figure 9). These beaches are often steep with coarse, poorly sorted sediments on more exposed coasts (Nordstrom, 2000), but can also accumulate fine or even muddy sediments in some settings (Martin et al., 2005). Salient beaches can block the



Figure 10 Beach loss seaward of coastal armoring that includes a seawall and revetment on exposed coast in Pacifica, California, USA. Note the lack of dry sand zones and lateral access for beachgoers.

littoral transport of sediments resulting in significant erosion to downdrift beaches (Thomalla and Vincent, 2003). For example, on an exposed Atlantic coastline in the United Kingdom, a series of offshore breakwaters caused the disappearance of the longshore bar and trough system, altering the surf zone and allowing higher waves to reach the shore between the breakwaters, eroding the beaches and creating a need for beach filling (Thomalla and Vincent, 2003). However, studies that have compared the effects of low-crested breakwaters on a variety of sedimentary habitats have suggested that dissipative beaches tend to be particularly severely affected by these changes, especially where riverine inputs lead to accumulation of fine sediments and organic matter, creating stagnant conditions typical of lagoons (Martin et al., 2005). Detached offshore structures are not, however, common in the more sheltered waters of estuaries.

8.02.4.2 Ecological Impacts of Structures

Despite the use of coastal armoring on coastlines around the world for thousands of years, numerous studies of the physical effects, costs and efficacy, and a very active debate on the geomorphic impacts of these structures on open and sheltered coasts, the ecological effects of these structures have been little studied and are poorly understood. A recent review that focused on sheltered coasts by the Ocean Studies Board of the US National Research Council stated that remarkably little is known about the effects of coastal defense structures on native coastal habitats and their communities, nor how they change ecosystem functions and services provided by natural ecosystems or introduce new ones (NRC, 2007). Even less is known concerning the ecological effects of these structures on open-coast ecosystems, such as beaches (Dugan et al., 2008; Walker et al., 2008).

As a consequence of this lack of knowledge, ecological impacts have generally not been considered in policy decisions regarding coastal armoring. However, as human populations continue to flock to the coast, sea level rises and coastal erosion accelerates; the need to understand the ecological consequences of armoring, in all its forms, on coastal ecosystems is increasingly urgent. We review existing case studies and information to synthesize the current understanding of the ecological impacts of armoring and coastal defense structures and identify urgent research needs. Major themes of our review include ecological effects of (1) the loss of habitat and alteration of processes in soft-sediment shores and benthos and (2) the creation of artificial and novel hard substrata in predominately soft-sediment ecosystems.

8.02.4.2.1 Loss of habitat

When the footprint of a man-made coastal structure covers and directly reduces existing habitat, the magnitude of loss of coastal habitat, known as "placement loss", varies with the type and construction of the structure, as well as its location on the shoreline and the characteristics of adjacent habitats. For example, revetments, rock groynes, and jetties with broad foundations cause more habitat loss per unit of height than do structures, such as seawalls or bulkheads with more vertical profiles (Griggs, 2005b). Structures placed adjacent to soft sediments are likely to have much larger impacts on this adjacent habitat than would be the case if the adjacent habitat was rocky reef.

In estuaries and bays, shoreline development and modifications associated with urbanization have exerted major impacts on both the area and the quality of natural habitats (Short and Burdick, 1996; Allen, 2000; Kennish, 2002; Zaikowski et al., 2008). When infrastructure and armoring cover and replace shoreline and marsh vegetation, they reduce water filtration, ecosystem functions, and connectivity among habitats. Shoreline armoring, especially bulkheads and seawalls, steepens shorelines, eliminates intertidal habitats, reduces structural complexity, and increases nearshore depths, thereby reducing or eliminating valuable shallow-water nursery and refuge habitat for many estuarine species (Peterson et al., 2000; Bilkovic et al., 2006; Seitz et al., 2006; NRC, 2007; Toft et al., 2007; Bilkovic and Roggero, 2008). This loss of intertidal and shallow-water estuarine habitats, including salt marshes and seagrass beds, to waterfront development, armoring, and infrastructure has been severe in many regions (Duarte, 2002; Seitz et al., 2006). In addition, the deepening and narrowing of tidal channels resulting from armoring, channelization, and coastal infrastructure have been associated with increased stratification and hypoxia in urbanized estuaries (Zaikowski et al., 2008).

For sandy beaches, although a large number of studies have quantified the responses of beach widths and profiles to a great variety of forms and applications of coastal armoring, they do not account for the relative responses of the different ecological zones of the beach habitat (e.g., McLachlan and Jaramillo, 1995), further limiting the understanding of ecological impacts. A conceptual framework developed for open-coast beaches proposed that a number of ecological impacts of armoring may be predicted using changes in the widths of different ecological zones of the beach as proxies for habitat loss (Figure 11) (Table 1) (Dugan and Hubbard, 2006). As the width of the overall beach and intertidal zone becomes narrow from the effects of placement loss and passive erosion in front of armoring structures, habitat area is lost disproportionately from upper shore zones. Thus, the effects of armoring are predicted to be

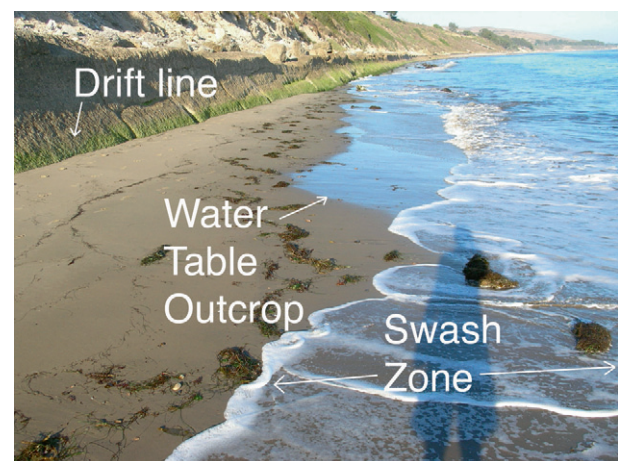


Figure 11 This view looking east along an old concrete seawall on the Gaviota coast of California, USA, at low tide illustrates the attenuation of ecologically important intertidal zones on a beach seaward of coastal armoring. Adapted from Dugan, J.E., Hubbard, D.M., 2006. Ecological responses to coastal armoring on exposed sandy beaches. *Shore and Beach* 74 (1), 10–16.

Table 1 Hypotheses concerning ecological effects of alongshore armoring on beaches

As beach width narrows in response to armoring structures:

- Upper intertidal, supralittoral, and coastal strand zones are lost disproportionately.
- Loss of drier upper beach zones decreases number of habitat types available and room for migration of habitats/zones and macroinvertebrates with changing ocean conditions.
- Reduction in habitat types reduces diversity and abundance of macroinvertebrates.
- Loss of upper beach habitat eliminates nesting habitat for sea turtles, fish, birds, etc.
- Lack of dry sand habitat and increased wave reflection associated with structures alter deposition and retention of buoyant materials (e.g., macrophyte wrack and driftwood), further affecting upper shore biota and processes, including nutrient cycling.
- Intertidal predators, such as shorebirds, respond to the combination of habitat loss, decreased accessibility at higher tides, and reduced prey resources.

greatest and occur earliest on the landward-most coastal strand (e.g., Feagin et al., 2005; Dugan and Hubbard, 2010) and supralittoral dry sand zones. Habitat near the drift line, the primary zone for wrack-associated invertebrates, may also be greatly reduced or eliminated. As the drift line habitat shifts from the beach to the armoring structure, rich three-dimensional infaunal beds characteristic of this zone are replaced with steep, reflective, two-dimensional artificial hard substrata.

Results of the few comparative studies of armored and unarmored beaches to date support these predictions. This general framework with some modification may also apply to the shores of bays and estuaries but has yet to be specifically examined. The scale of habitat and ecological effects of armoring was observed to be strongest for the upper shore zones in studies of seawalls along an undeveloped open coastline in California (Table 2) (Dugan and Hubbard, 2006; Dugan et al., 2008) and in the sheltered waters of Puget Sound (Sobocinski et al., 2010). In the open-coast study, there were no high beach zones (above the drift line) on the armored segments compared to adjacent unarmored segments where they averaged 3.5 m in width (Dugan and Hubbard, 2006; Dugan et al., 2008). This was consistent with the scale of placement loss expected for the seawalls studied and demonstrated the relative ecological importance of this impact on narrow beaches. The overall narrowing of the beach observed above the water table outcrop on armored segments, averaging 11.4 m, was, however, much greater than that expected from placement loss, suggesting the effects of passive erosion. In contrast to these results demonstrating armoring effects on the widths of mid-beach zones, Jaramillo et al. (2002b) found no significant differences in beach widths for a newly constructed seawall in an open-coast setting. This may be related to the differences in age of the seawalls in the two studies (20 months vs. 60+ years). However, no comparisons were possible for upper beach zones because the Jaramillo et al. study did not compare zone widths above the drift line.

An important consideration relative to the generality of these predictions is the location of the armoring structure on the beach profile, which affects the amount of interaction with waves and tides and the resulting physical impacts (Weigel, 2002a, 2002b, 2002c). Habitat loss is expected to scale with the intensity of interaction between structures and coastal processes (e.g., wave reflection and tidal action) which is predicted to increase as the structure ages and as sea levels rise. The ecological impacts of any armoring structure would be expected to respond similarly, whether location on the beach profile is due to initial placement or subsequent erosion of the shore.

Loss of hard natural habitat, such as rocky reefs, has generally been considered to be less of a problem (Thompson

Table 2 Average scale of ecological effects of armoring detected for open-coast beaches expressed as the ratio of mean values for pairs of unarmored and armored beach segments

<i>Ecological characteristic</i>	<i>Scale of effect</i>
Intertidal zone widths	
Upper beach limit to drift line	36× ^{***}
Upper beach limit to WTO	2.1× ^{***}
Macrophyte wrack (standing crop) ^a	374× [*]
Macroinvertebrates (upper shore)	
Species richness	n.d.
Abundance	10.6× [*]
Biomass	16.1× ^{***}
Mean individual size	1.6× [*]
Shorebirds	
Species richness	2.0× ^{***}
Abundance	3.7× [*]
Gulls	
Species richness	2.0× [*]
Abundance	4.8× ^{***}
Other birds	
Species richness	3.3× ^{***}
Abundance	7.7× ^{***}

^{*} $p \leq 0.05$ ^{**} $p \leq 0.01$ ^{***} $p \leq 0.001$

n.d. not detected; WTO, World Trade Organization.

^aValues from Dugan, J.E., Hubbard, D.M., 2006. Ecological responses to coastal armoring on exposed sandy beaches. *Shore and Beach* 74 (1), 10–16; and Dugan, J.E., Hubbard, D.M., Rodil, I.F., Revell, D., 2008. Ecological effects of coastal armoring on sandy beaches. *Marine Ecology* 29, 160–170.

Note seawalls in the study were >60 years old and interacted daily with high tides.

et al., 2002), because of the idea that artificial structures may act as suitable surrogate habitat. Although this may be true for some subtidal taxa, which settle in similar amounts on natural and artificial structures (Glasby, 1999; Chapman and Clynick, 2006), it is not necessarily true for intertidal (Chapman, 2003) or supratidal (Attrill et al., 1999) habitats, nor for fish living in adjacent waters (Able et al., 1998). This is discussed in greater detail in a subsequent section.

Overall, our review suggests that loss of coastal habitat caused by alongshore armoring structures affects upper shore, intertidal, or shallow-water zones disproportionately with greatest relative loss or elimination of habitats evident higher on the shoreline. Critically, this includes the loss of key ecotonal and transitional habitats between land and sea, such as coastal

strand, dune, salt marsh and other vegetated zones, supratidal and high intertidal zones, and shallow-water habitats on armored shores in estuaries, bays, and beaches. Loss of these key habitats will cause significant changes in biodiversity and community composition, altered ecosystem function, processes and services, and reduced connectivity between terrestrial and aquatic habitats of these important coastal ecosystems. Effects on habitat also appear to be greater for soft sediments than for hard substrata, but how much these apparent differences are due to real differences in the impacts versus a bias in the intensity of research in different habitats has yet to be determined.

Conceptual predictions of soft-sediment habitat loss that extend beyond placement loss for shore-normal structures, such as groynes and jetties, and for detached breakwaters and combinations of these structures are more elusive. These structures can alter benthic communities by creating novel habitats that either are unusually sheltered from waves or lack shallow or transitional habitats and gradients in depth. Habitat loss in the form of reduced beach widths can occur downcoast of a groyne or jetty where beach erosion is accelerated from the interruption of longshore sediment transport caused by the structure. At the same time, accretion of sediments upcoast of these structures can result in increased beach widths and habitat (Nordstrom, 2000). Similarly, creation and loss of inshore beaches can be associated with detached breakwaters (Thomalla and Vincent, 2003).

8.02.4.2.2 Alteration of ecological structure, function, and integrity

8.02.4.2.2(i) Coastal vegetation

Intact coastal vegetation, including mangroves, salt marshes, seagrasses, macroalgae, and coastal strand and dunes, buffers shores and retains sediments from the effects of erosive processes, such as tides, waves, and storms. These communities provide valuable ecosystem functions including primary production, water filtration, uptake of nutrients, detrital production, and degradation and carbon fixation (Costanza et al., 1997). Shoreline vegetation is often lost from open and sheltered habitats as bulkheads and seawalls both directly alter habitat and prevent the migration of the shoreline in response to the changing sea level. In the United Kingdom, for example, current ongoing losses of 100 ha yr^{-1} of coastal salt marsh have been estimated, due to the combined effects of erosion, reduced sediment inputs, land subsidence, and coastal defense measures (UK Biodiversity Group, 1999; Hughes and Paramor, 2004). Harmsworth and Long (1986) suggested that erosion from the seaward edge and prevention of landward migration because of seawalls could eradicate salt-marsh vegetation in a large British marsh. The loss of upper shore estuarine habitat, specifically elimination of a high-diversity vegetative transition zone, in front of seawalls has been associated with reduced diversity of salt-marsh plant communities (Bozek and Burdick, 2005). Shading of intertidal and nearshore habitats by coastal forests may provide cover, modify water temperatures, and create favorable microclimates for benthic and pelagic fauna (NRC, 2007). Nearshore vegetation may also serve as a source of terrestrial inputs to shallow waters and shoreline habitats (Sobocinski et al., 2010).

On open coasts, coastal strand vegetation is important in the formation of hummocks that can become embryo dunes and foredunes. This pioneering vegetation can be lost from armored beaches (Dugan and Hubbard, 2006). The effects of

human activities, such as beach grooming and trampling, coastal erosion, and sea-level rise on this already-restricted habitat (e.g., Feagin et al., 2005; Dugan and Hubbard, 2010), combined with the aggravated impacts caused by armoring, bode poorly for the survival of the coastal strand zone on coastlines that are either retreating or developed or both.

In marshes, the restriction of tidal flow and influence exerted by breakwaters, groynes, revetments, and other structures can lead to increased freshwater influence and the loss of tidal marsh species and function (NRC, 2007). Such freshening may also allow invasive and weedy species, such as *Phragmites*, to establish and outcompete native vegetation in coastal wetlands (King et al., 2007). Many seawalls in urbanized estuaries, such as Sydney Harbour, are also sites for storm water and urban runoff, usually channeled through few and often very large pipes. The continual influx of small amounts of freshwater can have small, but permanent, effects on intertidal assemblages, while large storm events may have larger effects, depending on the amount of tidal flushing.

8.02.4.2.2(ii) Land-sea connectivity

Shorelines are vital transitional zones linking terrestrial and marine realms (Polis and Hurd, 1996). Coastal armoring can sever these connections, reducing or eliminating key exchanges and functions, including organic and inorganic material transfers (detritus, nutrients, prey, and sediments), water filtration, and nutrient uptake (Bilkovic and Roggero, 2008). Shore-parallel armoring disrupts connections between the shoreline and the shallow water to terrestrial sources of sediments, such as coastal dunes, which may affect sediment dynamics and supply (Nordstrom, 2000). Although there has not yet been extensive research on this topic, the severing of the connection between the sea cliff erosion and the beach by coastal armoring may also constitute a significant reduction in sediment supply in some regions (e.g., Runyan and Griggs, 2003). Where estuarine shorelines are armored with impermeable bulkheads, the connectivity and input of allochthonous carbon from the marsh to the tidal waters and the benthos may be greatly reduced. The resulting reduction in food supply could impact benthic food webs, particularly for deposit-feeding infauna, such as the bivalve *Macoma balthica* (Seitz et al., 2006). Vegetated estuarine shorelines, including riparian forest and marsh, provide vital cover and detritus for terrestrial insects which are prey of fish, including juvenile salmonids (Levings, 1991).

8.02.4.2.2(iii) Wrack

Along with habitat loss, the alteration of physical processes that affect the deposition and retention of sediments on armored coasts may also affect the deposition and retention of buoyant material, including macrophyte wrack, driftwood, and other natural allochthonous debris, which can be important to biota as food or habitat (e.g., Colombini and Chelazzi, 2003; Dugan et al., 2003; Table 1). The significant relationship between wrack abundance and dry beach width found on California beaches (Revell et al., 2011) suggests that when dry upper beach zones are narrow or absent, wrack accumulation and its availability to beach consumers, microbial processing, and remineralization are greatly reduced. This prediction is supported by recent studies of open-coast beaches of California (Table 2; Dugan and Hubbard, 2006) and protected

beaches of Puget Sound (Sobocinski et al., 2010) that reported significantly lower standing stocks of wrack and driftwood on armored beaches compared to natural beaches. This effect of armoring could significantly impact the function of open-coast beach ecosystems in coastal nutrient cycling and dynamics (Dugan et al., 2011).

By contrast, increased macroalgal wrack deposition was associated with offshore breakwaters at some beaches in Europe due to increased shelter from waves (Martin et al., 2005). Groynes could also potentially trap higher accumulations of macrophyte wrack and terrestrial detritus delivered by littoral currents in accreting areas while reducing these organic inputs in the eroding areas, causing a variety of potential impacts. In some very sheltered estuaries in Australia, seagrass wrack accumulates in large amounts in intertidal zones adjacent to seawalls, killing all infauna (M.G. Chapman, unpublished data). The inability of wrack to move upshore on armored shores (Bozek and Burdick, 2005) may also affect survival of salt-marsh plants in areas where the wrack appears to ameliorate harsh conditions for developing plants (Chapman and Roberts, 2004).

8.02.4.2.2(iv) Benthic fauna

The loss of ecological zones, structural complexity, and habitat types associated with armoring could be expected to directly affect the diversity and abundance of intertidal and subtidal benthic fauna of sheltered and open coastlines. On open coasts, impacts of shore-parallel armoring are not well studied and only one community-level intertidal analysis has been completed to date for sandy beaches. Although more research is needed, the prediction of impacts of armoring to upper shore biota is generally supported by results of recent intertidal studies of beaches. For open-coast beaches in California, the abundance and biomass of mobile upper shore invertebrates were significantly greater on unarmored beach segments than on armored segments (Dugan et al., 2008; Table 2). For protected shores, results to date are similar. The abundance of

talitrid amphipods and insects was also significantly higher on natural beaches than on armored beaches for protected shores in Puget Sound (Sobocinski et al., 2010). Toft et al. (2007) reported a similar decline in high-shore benthos, which he suggested may have been due to a reduction in food related to the loss of shoreline vegetation. Upper beach invertebrates, such as talitrid amphipods, were not analyzed separately in a study of effects of a newly constructed seawall on the intertidal community by Jaramillo et al. (2002b) in Chile. On beaches in Australia, the densities of burrows of the upper shore scavengers, ghost crabs, were substantially lower on beaches where a seawall replaced dune habitat (Lucrezi et al., 2009) and on urbanized beaches with seawalls than on reference beaches (Barros, 2001). These results are consistent with a response to a loss of upper shore habitat.

On open-coast beaches, the distribution and survival of mobile invertebrates of the lower shore (e.g., donacid bivalves, whelks, isopods, and hippid crabs) may also be reduced by loss of habitat, changes in habitat quality, and restrictions on tidal migration, as well as the reduced availability of alternative sandy habitats (Klapow, 1972; McLachlan et al., 1979; Jaramillo et al., 2002b) imposed by seawalls. For example, the restriction of tidally generated landward migration of a cirrolanid isopod, *Excirolana chiltoni* imposed by a seawall, was illustrated by Klapow (1972) (Figure 12). Changes in suspended sediment concentrations and altered littoral current velocities and sediment transport rates in front of seawalls (Miles et al., 2001) could also affect the distribution and condition of benthic animals. Further studies of responses of mid- and lower beach biota to shoreline armoring are needed to evaluate if impacts observed for upper intertidal fauna also extend to the mid-intertidal and swash zones of open-coast beaches. Results of recent surveys in Chile and California suggest, however, that they do (Dugan, Hubbard, Jaramillo, Duarte, unpublished). However, the only Before-After/Control-Impact (BACI) study of short-term responses (20 months) of intertidal invertebrates to a newly constructed seawall on an

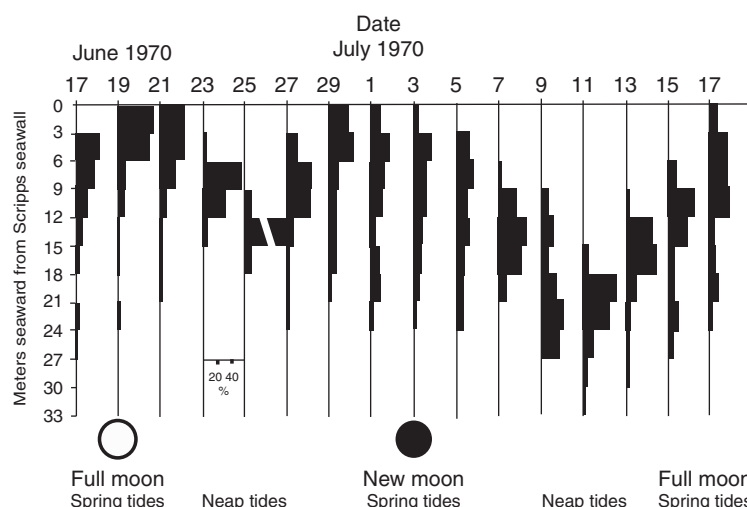


Figure 12 Semimonthly changes in the position of the intertidal zone occupied by the mobile beach isopod, *Excirolana chiltoni*, seaward of a seawall on the beach at Scripps Institution of Oceanography, San Diego, California during summer 1970. On spring tides, especially in June and early July, the wave wash interacted with the seawall causing truncation of the upper zone of the isopods as it literally hit the wall (e.g., 17, 19, 21, and 29 June). As the beach accreted after mid-July, the effect of the seawall on the distribution of the isopods was muted. From Klapow, L.A., 1972. Fortnightly molting and reproductive cycles in the sand-beach isopod, *Excirolana chiltoni*. Biological Bulletin 143, 568–591.

open-coast beach by Jaramillo et al. (2002a) did not find significant effects of armoring on the overall macroinfaunal invertebrate community, nor on populations of two abundant mobile invertebrates (a cirrolanid isopod, *Excirolana hirsuticauda*, and an anomuran decapod, *Emerita analoga*) that inhabit mid- to lower intertidal zones. This result suggests that additional factors, including the age and the position of the structure on the beach profile, may be important in predicting both habitat loss and ecological impacts. More information is anticipated, as Jaramillo and his students are now conducting studies on the same beach, >10 years after the construction of the seawall to evaluate ecological responses to armoring.

In one of the only investigations to date of the ecological effects of shore-perpendicular structures on soft-sediment intertidal communities on a sandy beach, Walker et al. (2008) demonstrated that groynes were associated with increased spatial variation in the structure of the macroinvertebrate assemblage. This study of a relatively small structure placed on an open-coast beach in Australia ~27 years earlier found that the groyne altered local hydrodynamic processes, which resulted in changes to the morphodynamic properties of the habitat. Erosion was evident on the side exposed to the predominant swell and waves, and accretion on the opposite side of the groyne. These local (<20 m) modifications to the accretion/erosion dynamics of the beach resulted in markedly coarser sediment in the eroding areas near the groyne and finer sands where the structure promoted beach accretion. This alteration in conditions caused changes in the invertebrate communities on the beach: (1) in areas close to the wall, there were consistently more species recorded in accreting areas (18 species) than in eroding ones (13 species) and (2) faunal assemblages differed significantly in terms of their species composition and abundance between the two sides of the wall, particularly close to the structure (Analysis of Similarity (ANOSIM) proximal locations $R = 0.51$, $p = 0.029$) (Figure 13).

In many parts of the globe, armoring of coastal areas with groynes is far more extensive, with numerous structures placed in close proximity (Figure 6) (Bush et al., 2001; Fanini et al., 2009) and potentially greater ecological impacts. Walker et al. (2008) showed the impact on the distribution and abundance of the beach macrofauna was limited to within 10 m of a single groyne during their study although if the groynes were placed in close succession, it was predicted they would have a substantial impact on both the physical characteristics and the macrofaunal communities of the beach, potentially affecting much larger areas of the coastline. In some areas, groynes may, by contrast, provide suitable habitat for intertidal species that live on hard substrata (Pinn et al., 2005; see later discussion).

In estuaries, the predicted ecological impacts on sub-tidal benthic fauna from the loss of shallow-water zones and depth gradients associated with shoreline armoring and bulkheads are supported by the results of recent studies. Ecological thresholds for biotic indices of nearshore macrobenthic communities were reached when the amount of developed shoreline in estuaries exceeded 10% in an analysis of Chesapeake Bay by Bilkovic et al. (2006). The density and diversity of subtidal benthic bivalves, which make up >50% of benthic prey biomass, were highest along natural marsh shorelines compared to riprap or bulkhead-armored shores in Chesapeake Bay, whereas overall infaunal density and diversity were highest along natural marsh and riprap shores (Seitz et al., 2006).

On a small scale, bulkheads and levees eliminate or significantly reduce access to intertidal marsh habitat, but these effects can accumulate to a larger area of impact, fragmenting habitat and reducing connectivity (Peterson and Lowe, 2009). Partyka and Peterson (2008) found that even the small patches of marsh habitat supported a greater diversity of fauna than nearby restricted habitats. They suggested that the relative quality of marsh-edge habitat depends upon the surrounding landscape and that ecosystem health is affected strongly by the spatial arrangement of the marsh and human alterations of the shoreline.

An extensive review of the responses of infauna and mobile biota to a number of shore-parallel offshore coastal defense structures (characterized as low crested) on the coasts of Spain, Italy, and the United Kingdom by Martin et al. (2005) showed that impacts were highly variable from place to place. However, they reported differences in soft-bottom characteristics and biotic communities relative to control sites, primarily landward of the structures where wave-sheltered conditions were created (Martin et al., 2005; Bertasi et al., 2007). A general increase in infaunal invertebrate species richness observed in the vicinity of the structures was primarily related to the presence of new species settling on the hard structure itself (Moschella et al., 2005) and the colonization of lagoonal and quiet water species in the sheltered conditions landward of the structures (Martin et al., 2005). Increased accumulation of fine sediments, silt, and organic matter landward of the structure was generally associated with these faunal changes and for a few of the sites, anoxia and faunal impoverishment were observed in this zone.

In some regions (e.g., the Mediterranean), coastal defense structures are often used in combination with beach filling or nourishment. The effects on shallow benthic fauna related to the use of extensive beach fills in combination or not with hard defense structures, have been studied along about 50 km of coasts in the North Adriatic sea (Colosio et al., 2007). This study has shown that when nourishment is applied in combination with breakwaters, some of the impacts related to the addition of sediments on nearshore habitats and assemblages can be mitigated by the increased stability of the sediments; however, these benefits are largely canceled out by the direct impact of the breakwaters, which create artificially sheltered conditions that enhance the deposition of very fine sediments and attract nonnative assemblages.

8.02.4.2.2(v) Fish and nursery habitat

Estuaries and their complex mosaic of habitats and resources can be important for the survival of early life stages of numerous species of fish and crustaceans, many of which support commercial fisheries or are ecologically important as prey for higher trophic levels (Able et al., 1998, 1999; Peterson et al., 2000; Peterson and Lowe, 2009). The modification of subtidal habitats by coastal development can alter biodiversity, distributions, trophic interactions, community assemblages, and the quality of nursery habitat of the estuarine ecosystem (Able et al., 1998; Toft et al., 2007). Evidence that armoring of estuarine shorelines significantly impacts the distribution and abundance of fish and nekton is accumulating from a number of regions, but results are not all in agreement. Abundance of fish, crabs, and shrimp and nekton diversity were lowest on shorelines altered with bulkheads and rubble and highest along pristine shorelines in estuaries on the Gulf of Mexico

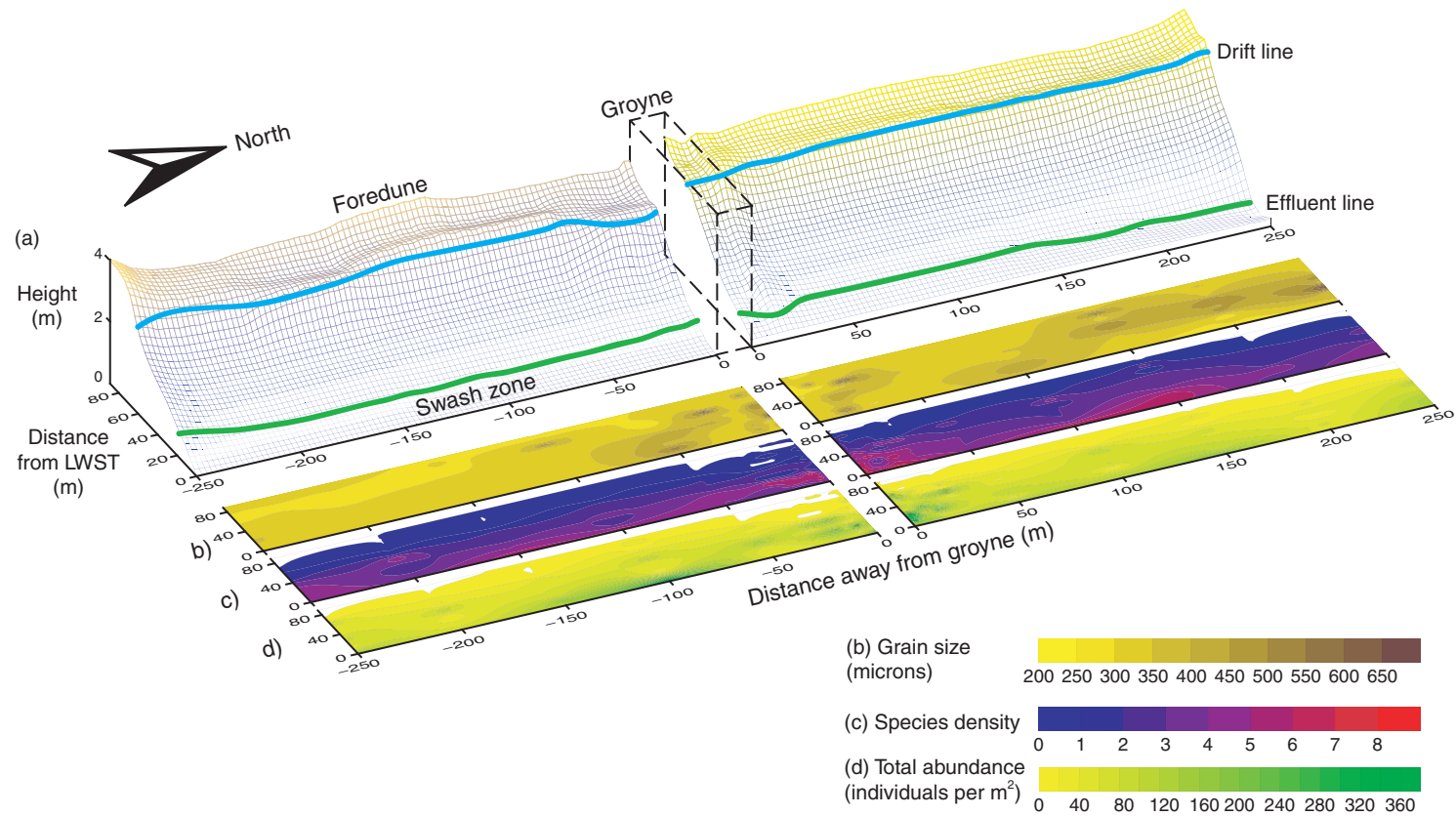


Figure 13 Beach profile of (a) the areas surveyed 250 m on either side of a groyne, and contour maps showing the impact of a groyne on (b) mean grain size (microns), (c) average species density (number of species per sample), and (d) total macrobenthos abundance (mean number of individuals per m²). Modified from Walker et al., 2008.

(Peterson et al., 2000). In the Chesapeake Bay, reduced integrity of fish communities was associated with both upland development and bulkheads (Bilkovic and Roggero, 2008). Fish communities along developed shorelines and those with bulkheads were dominated by a few generalist species, while those with little upland development and natural or riprap shores were diverse, and included tidal marsh species (Bilkovic and Roggero, 2008). Epifaunal nekton richness (fish and invertebrates) was consistently higher adjacent to unrestricted shores than adjacent to restricted and hardened shores in a Gulf of Mexico estuary (Partyka and Peterson, 2008). In the Chesapeake Bay, the density and diversity of predators (fish and invertebrates) were highest along natural shorelines, while crab density was significantly higher in natural marsh habitats compared to those with bulkheads (Seitz et al., 2006).

By contrast, Able et al. (1998) showed no effect of shoreline development on fish assemblages, unless there was an extensive concrete cover completely shaded the habitat. Likewise Davis et al. (2002) showed very localized and patchy effects of armoring on fish populations in San Diego Bay (see also Martin et al., 2005). Clynick (2006) also indicated that marinas may support very large abundances of juvenile fish, perhaps due to the sheltered conditions. Many fish accumulate in large densities around urban infrastructure, such as jetties and piers (Pérez-Ruzafa et al., 2006), with some fish predominantly found on such structures (Clynick, 2008). This may be, in part, due to increased amounts of food living on the structures themselves (Clynick et al., 2007), but the speed with which fish arrive at artificial reefs (Baine, 2001) and other artificial structures introduced into marine habitats (Chapman and Clynick, 2006) suggests that they are not dependent on the food supply.

The extensive review of the responses of infauna and mobile biota to low-crested shore-parallel offshore coastal defense structures by Martin et al. (2005) indicated that fish species typical of rocky shores were also attracted to the structures, but these were primarily juvenile stages. The effects of alongshore and shore-normal armoring on surf zone fish and crustaceans of open-coast beaches have not been evaluated.

Some ecological effects of coastal armoring may involve interactions with human activities. Artificial structures that are easily accessible are popular locations for recreational angling, or for harvesting biota that grow on the structures (Airoldi et al., 2005b). Thus, groynes, jetties, marinas, and other similar structures that attract fish may lead to increased human harvesting pressure on these fish, possibly modifying food-web dynamics in nearby habitats. However, it remains unclear how fish populations are affected by these structures, particularly whether they result simply in attraction and concentration of fish or increased productivity.

8.02.4.2.2(vi) Barriers to movement of animals and wrack

Mobile scavengers, such as ghost crabs, talitrid amphipods, and isopods, occur in both the vegetated dunes landward of the backshore and the uppermost part of the intertidal and supratidal zone of the unvegetated shore or beach. The animals move between the dunes and the beach, mostly to feed on stranded material. Movement landward from the beach into the dunes is especially important for the intertidal part of the population during extreme weather events, when dunes serve as refuges (Christoffers, 1986). Seawalls placed at the back of the shore form a barrier to animal movement, preventing

access to dunes for intertidal individuals and vice versa, access to the beach for dune inhabitants (Lucrezi et al., 2009). Both effects will have negative consequences in terms of higher risk of drowning and displacement during storms and decreased food availability. Impacts of seawalls on animal movement between the beach and the dunes may also be evident for other taxa, such as small rodents and other terrestrial mammals, accessing beaches to feed on the strandline (Carlton and Hodder, 2003; Bird et al., 2004) as well as the chicks of nesting shorebirds (e.g., piping plover and snowy plover) which move to the intertidal shoreline for foraging from backdune or marsh habitat (Burger, 1994).

Groynes and jetties can also create barriers to the longshore movement of mobile benthic animals and propagules, particularly if arrayed in a series or in groyne fields along a coastline, as found along the coast of Italy (Bondesan et al., 1995; Cencini, 1998; Fanini et al., 2009). They could potentially trap higher accumulations of macrophyte wrack and terrestrial detritus delivered by littoral currents in the accreting areas, while reducing these organic inputs in the eroding areas, causing a variety of potential unintended impacts.

8.02.4.2.2(vii) Wildlife support

The support of wildlife species, including birds, turtles, and marine mammals, is a very important ecological function of coastal ecosystems (e.g., Schlacher et al., 2007). Beaches and estuaries provide valuable coastal habitat for foraging, roosting, and nesting avifauna, including shorebirds or waders, gulls, seabirds, and even a variety of land birds (Hubbard and Dugan, 2003; DeLuca et al., 2008). Loss of habitats used during migration, foraging, and overwintering has been implicated in the declines of populations of many species of shorebirds and is a major concern for shorebird conservation (Howe et al., 1989; Brown et al., 2001), as are the effects of climate change (e.g., Kendall et al., 2004). Shorebirds require abundant prey resources in order to meet their high metabolic rates and relatively high daily energy requirements (Kersten and Piersma, 1987). For example, declines in horseshoe crab eggs in heavily armored Delaware Bay have been associated with reduced body weights and abundance of migratory red knots (Niles et al., 2009). Shorebird diversity and abundance have been correlated with prey availability on California beaches (Dugan et al., 2003). Changes in habitat area, tidal availability, and quality and in intertidal prey availability resulting from armoring have the potential to negatively impact coastal avifauna.

For open-coast beaches, existing evidence, although limited, suggests that coastal avifauna can respond strongly to armoring. The significant differences found in the diversity and abundance of shorebirds, as well as seabirds and gulls, between armored and unarmored segments of narrow exposed beaches of California (Dugan and Hubbard, 2006; Dugan et al., 2008) suggested that ecological impacts on coastal avifauna can be substantial (Table 2, Figures 14 and 15). Of note, the significant effects of armoring on birds were observed during low-tide surveys when the greatest amount of intertidal habitat was available. During higher tides, bird use would be eliminated in front of the seawalls. The differences in shorebird abundance associated with coastal armoring (less than threefold) exceeded that predicted by the overall loss of beach habitat area from armoring (twofold) (Table 2), suggesting that other factors, including prey abundance, availability of high-tide feeding habitat and refuges,

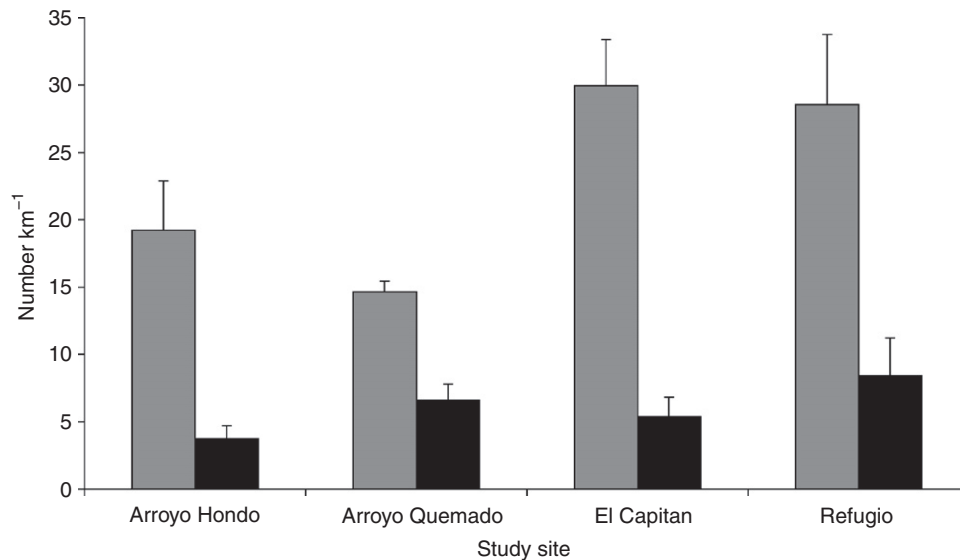


Figure 14 Mean abundance of shorebirds (+ one standard error, $n=8$) during fall migration for adjacent bluff-backed (gray bars) and armored (black bars) segments of coastline at four beaches in August and September 2005.

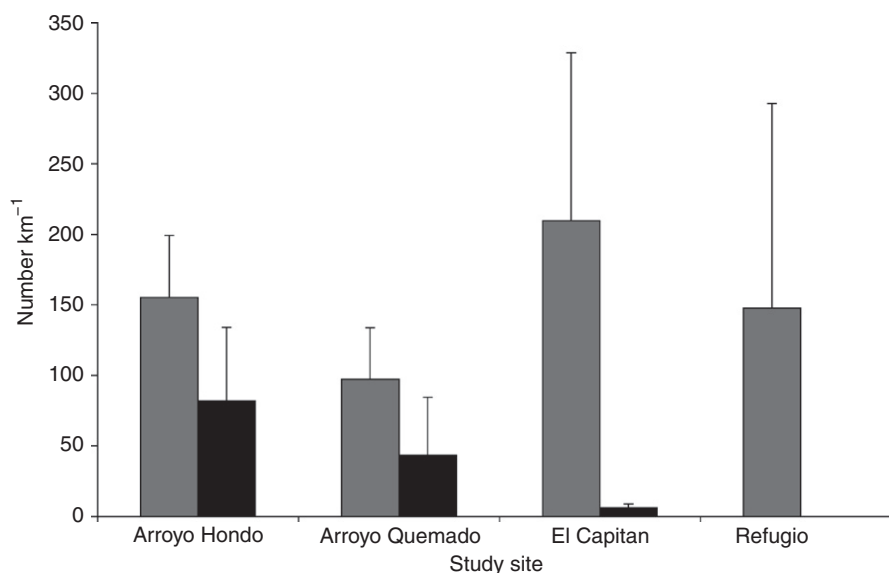


Figure 15 Mean abundance of gulls, cormorants, and herons during fall migration (+ one standard error, $n=8$) for adjacent bluff-backed (gray bars) and armored (black bars) segments of coastline at four beaches in August and September 2005 in Santa Barbara County, California. Average density of these birds was 25% lower on segments next to armoring structures.

as well as other landscape factors, may have contributed to the observed responses. Responses of gulls and other birds (both greater than fourfold) to armoring also exceeded that of the loss of habitat (Table 2), suggesting that armoring affects the quality of habitat needed for roosting or loafing by gulls and seabirds, in particular. The avoidance of seawall-backed beaches not only by foraging shorebirds (Figure 14), but also by roosting birds, such as gulls, pelicans, and other seabirds (Figure 15), indicates that impacts of armoring beyond food-web requirements need to be considered for coastal avifauna.

Beaches are critical nesting areas for sea turtles (Wood and Bjorndal, 2000) and a number of specialized species of fishes, including capelin *Mallotus villosus* (Nakashima and Taggart, 2002), grunion *Leuresthes tenuis* (Smyder and Martin, 2002),

surf smelt *Hypomesus pretiosus* (Rice, 2006), and sand lance *Ammodytes hexapterus* (Reeves et al., 2003), as well as horseshoe crabs, *Limulus polyphemus* (Jackson et al., 2008). These taxa require beach habitat that is above the reach of average high tides to successfully reproduce and incubate their eggs. The loss of upper beach zones resulting from armoring alters or eliminates nesting habitat for these animals (Moiser and Witherington, 2002; Rizkalla and Savage, 2011). The effects of armoring on the microclimate (temperature, humidity, and light intensity) of these nesting zones can also be important (Wood and Bjorndal, 2000; Jackson et al., 2008). For example, mortality of surf smelt embryos was 50% higher on beaches with bulkheads in Puget Sound, Washington (Rice, 2006). These impacts are of particular concern (Jackson et al., 2008) because sea turtles

are already threatened by a variety of other human activities; surf smelt are important prey for juvenile salmon, a threatened species (Rice, 2006), and horseshoe crab eggs are a key resource for migratory shorebirds (Sweka et al., 2007).

8.02.5 Coastal Infrastructure and Armoring as Novel Substrata for Biota

At the same time as coastal infrastructure and armoring structures occupy and alter soft-sediment habitats, they introduce new intertidal or subtidal hard substrata that were not previously available, particularly when placed in predominately sedimentary environments, creating opportunities for animals from other habitats to colonize new areas. Despite the preceding list of impacts that have shown decreases in diversity of associated flora and fauna, at a first glance, coastal infrastructure and armoring seem to create suitable habitat for many marine organisms which rapidly settle and spread on the new hard substrata. It is precisely because of this trend that waste material is often dumped at sea to create artificial reefs. There have, however, been relatively few studies of the value of such reefs for species other than fish (see reviews by Baine (2001) and Svane and Petersen (2001), and recent work by Perkol-Finkel et al. (2006), Miller et al. (2009) and Burt et al., (2009)). Similarly, there have been relatively few studies of the value of armoring and urban infrastructure as habitat for marine fauna and flora, although many subtidal epibiota are fouling species rapidly colonize artificial structures (Glasby and Connell, 1999; Chapman and Clynick, 2006). Some structures, such as floating buoys and pontoons, create novel habitat for which there are no natural equivalents (Connell, 2000), whereas other surfaces, for example, subtidal walls, may be closer in morphology to natural cliffs and rocky reefs and have similar biotic assemblages (Glasby, 1999). Nevertheless, as described previously, although many species of fish aggregate around coastal infrastructure, such as marinas and wharves, these assemblages can be reduced or consist of a different mix of species than occurs on natural reefs, depending on the type of habitat created by the artificial structures (e.g., Able et al., 1998, 1999; Rilov and Benayahu, 1998).

Although artificial structures are often uncritically claimed as reasonable mimics of natural rocky and biogenic reefs, there is growing evidence that human-made artificial structures do not function as natural rocky or biogenic habitats. Indeed, numerous studies document changes to the assemblages of species inhabiting such structures (Connell, 2000; Chapman and Bulleri, 2003; Bulleri et al., 2005; Moschella et al., 2005), local loss of species of particular functional groups, for example, large grazers and predators (e.g., Chapman, 2003), low species and genetic diversity (Johannesson and Warmoes, 1990; Chapman, 2003; Fauvelot et al., 2009), and the presence of flora and fauna that often represent an early stage of succession, dominance by opportunistic and invasive species (Russell, 2000; Bacchiocchi and Airolidi, 2003; Bulleri and Airolidi, 2005; Glasby et al., 2007), and different ecological interactions (Iveša et al., 2010; Klein et al., 2011) and functions (Bulleri, 2005a; Moreira et al., 2006; Miller et al., 2009; Martins et al., 2009; Perkol-Finkel and Benayahu, 2009). Even in the comparatively rare situations when artificial structures have been specifically designed to mimic natural habitats and enhance species of recreational, commercial, or naturalistic value (e.g., artificial reefs), there has

been no consistent evidence that these aims have been achieved (Svane and Petersen, 2001; Perkol-Finkel et al., 2006; Burt et al., 2009; Miller et al., 2009).

Intertidally, the ecological value of shorelines that have been altered to create new hard substrata appears to be quite low. This appears to be due to a combination of three major characteristics of the artificial surfaces themselves. In areas where natural shores are gently sloping, the steep vertical surfaces of most types of infrastructure provide a much smaller extent of intertidal habitat, perhaps reducing the intertidal area extent from low to high water from tens of meters to only a few meters (Chapman, 2003). This is likely to reduce the numbers of species via species–area relationships alone. In addition, when the resident species are more suited to living on gentle slopes, they may not be able to survive on vertical surfaces, especially where there is a great deal of wave action. Therefore, differences in intertidal slope may affect the amount of available intertidal area in addition to its quality. Second, when the material used to create the infrastructure is different from that of natural habitat, this may in itself affect settlement or survival of species (Davis et al., 2002; Moreira, 2006). Third, and possibly most important, the artificial surfaces of most armoring infrastructure lack many of the microhabitats found on natural rocky shores, for example, shaded crevices, rock-pools, etc. Many of the species that have been documented as being absent from seawalls, for example, in Sydney, Australia, are species that use these microhabitats (Chapman, 2003). Finally, artificial structures are characterized by unnaturally high levels of disturbance from both natural (e.g., storms and sediment scour) and anthropogenic (e.g., harvesting, trampling, and maintenance works) sources, which tends to favor the establishment of species with opportunistic traits (Airolidi et al., 2005a; Bulleri and Chapman, 2010).

Although correlations between the structure of artificial habitats and species composition have been reported, correlations cannot be used to imply causation. The importance of the structural simplicity of such habitats in determining the composition of species that can live on them has been demonstrated in only few experimental studies designed specifically to test such hypotheses. Thus, Chapman and Blockley (2009) demonstrated that creating artificial rock pools into a vertical seawall (Figure 16) increased the diversity of species



Figure 16 The construction of artificial rock pools in a new seawall being constructed in Sydney Harbour, Australia.

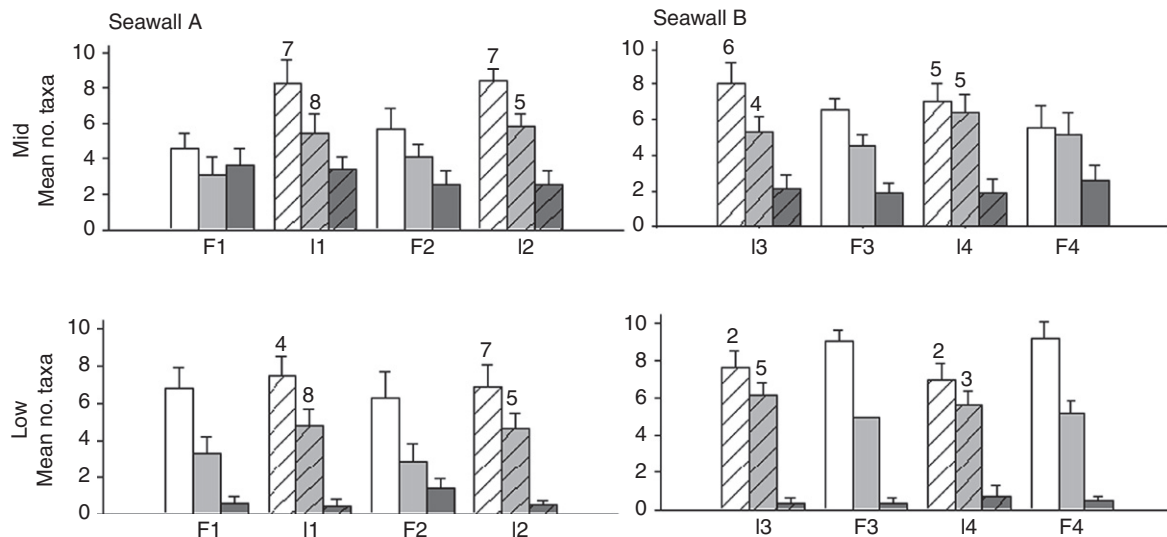


Figure 17 Mean (SE) number of taxa of algae (white), sessile animals (light gray), and mobile animals (dark gray) which settled on flush (clear) or indented (hashed) mortar at mid- and lowshore levels for each of four sites in two repaired seawalls, A and B. The sites were paired along the lengths of the repaired walls; F1/I1 ... F4/I4. There were 9 times of sampling on Seawall A and 7 times of sampling on Seawall B. Numbers above the bars show the number of times that the number of taxa in the site with indented mortar exceeded that in the adjacent site with flush mortar.

that colonized the wall more than threefold, both by creating shaded surfaces and by creating pools that retained water during low tide. Similarly, the simple provision of finger-deep indentations between adjacent stone blocks on intertidal walls, rather than building walls with mortar flush with the surfaces of the blocks (favored by planners because of its aesthetic appearance; M.G. Chapman, personal observation), can increase the diversity of species that settle on these surfaces (Figure 17).

Nevertheless, because many species are documented to live on or around armoring and urban infrastructure, it has been suggested that these artificial substrata may adequately represent natural habitats (e.g., Thompson et al., 2002; Pister, 2009) or may, in fact, compensate for loss of habitat elsewhere (e.g., Iannuzzi et al., 1996). Other authors have suggested adding more artificial structures to urban coastlines to create additional habitat (e.g., Iverson and Bannerot, 1984). This approach to conservation should, however, be treated with a great deal of caution without further research into the value of artificial substrata for survival of both common and rare species. There has been little research on this issue to date, specifically on infrastructure and armoring, and most attempts at mitigating habitat loss by adding hard substrata have shown the created artificial habitats to be very poor compared to natural habitats (e.g., Roberts, 1993; Perkol-Finkel et al., 2006; Burt et al., 2009; Miller et al., 2009).

8.02.6 Large-Scale Effects

Among the less recognized ecological impacts of urban infrastructure and armoring are large-scale alterations of coastal seascapes and related effects on the dispersion and connectivity in marine populations. In most instances, armoring and artificial structures are built for a variety of human uses in areas which otherwise have soft-sediment habitats. This results in the

fragmentation, degradation, and loss of native sedimentary habitats, with impacts on biodiversity, biotic communities and populations, wildlife support, and a range of ecosystem functions over large spatial scales (e.g., Airolidi and Beck, 2007; Schlacher et al., 2007). At the same time, these structures create stepping stones or corridors for hard-bottom species (Glasby and Connell, 1999; Dethier et al., 2003; Airolidi et al., 2005a), allowing the spread of species into areas where they would not occur naturally (Figure 18). Thus, one of the major consequences of the introduction of substrata and artificial habitats which are alien to the original coastal ecosystem is that they tend to attract and support species typical of hard substrata, irrespective of their origin. This may result in the expansion of a species range into areas adjacent to those naturally occupied (e.g., Johannesson and Warmoes, 1986; Davis et al., 2002), but all too frequently leads to the rapid expansion of documented invaders (Bulleri and Airolidi, 2005). In most instances, artificial hard structures are associated with unnatural expansions in the distribution of native and nonnative hard-bottom species (Glasby and Connell, 1999; Bacchiocchi and Airolidi, 2003; Bulleri, 2005b; Martin et al., 2005), although the effect of the new structures themselves are usually confounded with other disturbances, for example, pollution, dredging, etc. (Ruiz et al., 1997). In the Wadden Sea, for example, where hard substrata are naturally scarce, ~730 km of artificial structures (harbors, causeways, dikes, piers, breakwaters, etc.) have introduced ~2–4 km² of hard substrata, providing new opportunities for a variety of hard-bottom species that is otherwise rare or absent in such sedimentary environments (Reise, 2005). Along the open coast of Italy bordering the north Adriatic Sea, which is naturally devoid of rocky shores, >190 km of rock-armored structures, built mainly in the past 40 years (Bondesan et al., 1995), have introduced >1 km² of artificial hard substrata within natural sandy environments. These structures have been extensively colonized by hard-bottom organisms, with a prevalence of opportunistic and weedy species, including numerous nonindigenous species (e.g., the green alga *Codium*

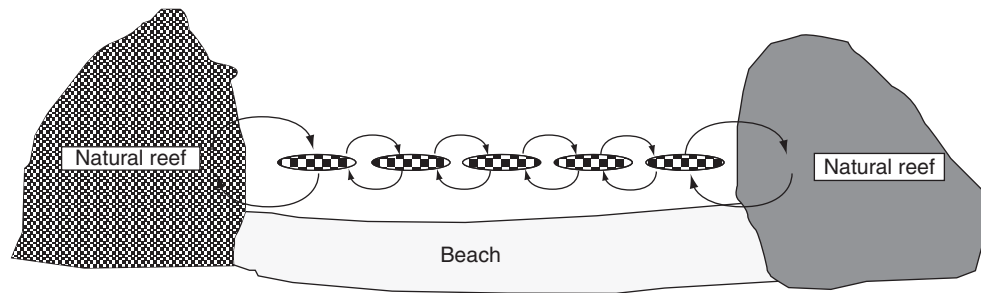



Figure 18 Conceptual diagram of the use of coastal armoring structures as stepping stones by species that require hard substrata. The figure, not to scale, illustrates possible interactions at a regional scale of tens to hundreds of kilometers. The proliferation of armoring (indicated as ) in areas with few natural rocky substrata can promote the dispersal of species outside their natural ranges, thus increasing connectedness between naturally isolated rocky reefs. Modified from Airolidi et al., 2005a.

fragile spp. *tomentosoides*, the Pacific oyster *Crassostrea gigas*, the predatory snail *Rapana venosa*, and a number of tunicates) (Bacchiocchi and Airolidi, 2003; Bulleri and Airolidi, 2005; Rius et al., unpublished data). In the Northern Gulf of Mexico, approximately 4000 oil and gas platforms have enhanced the dispersal of coral populations into areas where they were previously naturally absent (Sammarco et al., 2004). In fact, man-made structures may be particularly sensitive to invasions by nonindigenous species (Bulleri and Airolidi, 2005; Neill et al., 2006; Glasby et al., 2007; Vaselli et al., 2008; Dafforn et al., 2009), and experiments clearly point to the roles of severe disturbances and sheltered conditions which are typical of habitats with extensive amounts of artificial structures (Airolidi et al., 2005b; Bulleri and Airolidi, 2005).

While the impacts of increased habitat fragmentation and the resulting loss of connectivity have been broadly recognized and appreciated for many ecosystems, the potential consequences of enhanced connectivity have been little explored to date. On one hand, increased connectivity could provide new dispersal routes for threatened species among habitats, for example, by facilitating migration in response to climate changes (Hoegh-Guldberg et al., 2008). On the other hand, there is robust evidence to suggest that there could also be severe drawbacks, including the rapid expansion of weedy nonnative species (as discussed previously) or the breakage of natural barriers to distribution among isolated (e.g., by stretches of sandy or muddy habitats) and differentially adapted populations. For example, population genetic analysis on the limpet, *Patella caerulea*, from natural and artificial habitats at various sites along the Adriatic coast showed that genetic diversity (allelic richness and gene diversity) was significantly higher in populations inhabiting natural rocky shores than those on artificial structures (Figure 19) (Fauvelot et al., 2009). While the causes of these differences are not yet understood and require further investigation, they clearly suggest that the expansion of armoring and other structures may lead to genetic diversity loss in rocky shore populations at regional scales. Indeed, biotic homogenization is probably one of the major large-scale impacts expected from increasing urbanization (Sax and Gaines, 2003), but, despite important evolutionary consequences, the potential role of marine artificial structures in promoting genetic exchange remains virtually unexplored.

The potential drawbacks related to the expansion of non-indigenous species have received increasing attention in recent times. In fact, many alien species appear to settle or grow well

on marine artificial structures (e.g., Arenas et al., 2006; Locke et al., 2007; Tyrrell and Byers, 2007). Several recent studies indicate that marine man-made structures are particularly sensitive to invasions by nonindigenous species (Bulleri and Airolidi, 2005; Neill et al., 2006; Glasby et al., 2007; Vaselli et al., 2008; Dafforn et al., 2009). This is, of course, exacerbated when the artificial structures are associated with boating or shipping, for example, all of the major docks, ports, harbors, and marinas around the world, because the boats or ships themselves frequently continue the spread of exotic species, either via ballast water or by assemblages established on ships hulls. Therefore, introducing hard and sheltered substrata in such areas can clearly facilitate the spread of species that would otherwise have limited possibilities to further expand beyond the point of introduction. Some notorious examples are represented by the spread of the introduced green macroalgae, *C. fragile* ssp. *tomentosoides* (Bulleri and Airolidi, 2005) and *Caulerpa racemosa* (Vaselli et al., 2008) on breakwaters along sedimentary coasts of Italy. There is, however, mounting evidence that artificial structures could represent habitats that are intrinsically more vulnerable to invasions than natural habitats (Glasby et al., 2007). Experiments, for example, have clearly shown that the severe disturbances and sheltered conditions that, as discussed previously, are typical of artificial structures can be major drivers in facilitating species introductions (Airolidi et al., 2005b; Bulleri and Airolidi, 2005), by offering prolonged availability of unoccupied space or other resources. Recent work has also shown that colonization by nonindigenous epifauna could be enhanced on shallow moving substrata, such as floating docks (Dafforn et al., 2009), which has been interpreted as a consequence of the adaptation of species transported on ship hulls to resist high shear stress.

8.02.6.1 Effects on Adjacent Habitats

The ecological impacts of adding armoring and other infrastructure to shorelines can be found on reefs outside the immediate area of the structures themselves and beyond any impacts on the flora and fauna that actually occupy the structures. For example, large concentrations of fish around wharves or marinas might alter benthic assemblages by excessive grazing or predation (e.g., John and Pople, 1973), but such effects may be very small or patchy (e.g., Connell, 2001). Feeding by fish may assist in keeping associated benthic assemblage at an

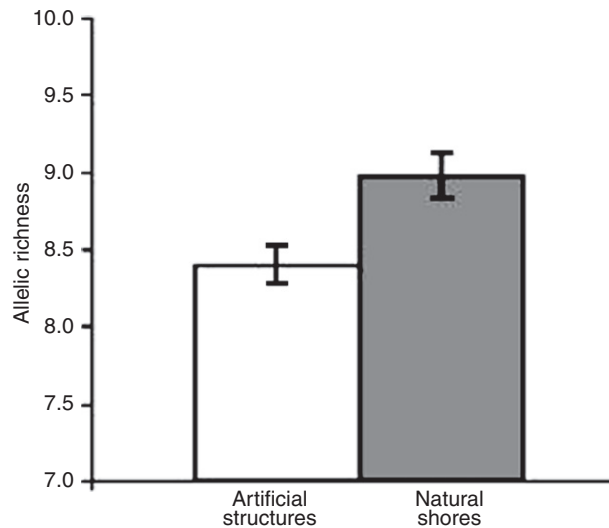


Figure 19 Average allelic richness of *Patella caerulea* on artificial structures and natural shores in the Adriatic Sea based on 14 samples and 12 diploid individuals. Modified from Fauvelot, C., Bertozzi, F., Costantini, F., Airoidi, L., Abbiati, M., 2009. Lower genetic diversity in the limpet *Patella caerulea* on urban coastal structures compared to natural rocky habitats. *Marine Biology* 156, 2313–2323.

early stage of succession (Carter et al., 1985) and, as described previously, the hard vertical shore edge created by walls may prevent movement of plant detritus either on- or offshore, creating an accumulation of plant material which may alter sediments and potentially infauna (Bozek and Burdick, 2005). Changes to current flow in nearby waters caused by these structures can create areas of stagnant water and hypoxia (Zaikowski et al., 2008), although, again, such effects may be quite variable and patchy (Martin et al., 2005), despite being locally very important (Airoidi et al., 2005a).

Other effects may be quite subtle, but potentially of great concern. For example, Goodsell et al. (2007) documented decreased diversity of intertidal species on natural rocky shores adjacent to seawalls, compared to those abutting other natural habitats, such as mangroves. Although there has been no further study of the processes that caused this reduced diversity, the shores adjacent to seawalls were both smaller and more separated from other rocky shores. These are both common characteristics of anthropogenically fragmented habitat, typically characterized by low biotic diversity. This suggests that intertidal rocky shores, although often naturally small and patchy habitats, may similarly be affected by habitat fragmentation by shoreline development as are terrestrial habitats.

More wide-scale impacts may arise from changes to ecological interactions, or reproductive output. Moreira et al. (2006), in the only study of effects of urban infrastructure on reproduction of species living on such structures, showed reduced reproductive output by a species of limpet, even though that species was found at high densities on seawalls compared to natural shores. This was likely due to the fact that the adult limpets on walls were all small, although whether this was due to reduced growth or increased mortality of large animals is not known. While the impacts of this effect may not be particularly severe when armoring and other infrastructure simply adds to natural rocky habitat, they may become very serious when such infrastructure replaces natural rocky shores, especially if it is argued that they are an effective substitute for natural habitats.

8.02.7 Potential for Recovery/Resilience

Because of the growing concern about effects of coastal armoring, there is considerable effort in finding an alternative to hard armoring for both estuarine and open shorelines, for example, 'living shorelines' (Living Shoreline Summit Steering Committee, 2006; Kabat et al., 2009). In estuarine settings, incorporating hybrid designs, which combine hard structures with natural elements, such as vegetation, woody debris, or shellfish reefs, might reduce negative impacts. Similarly, adding other habitat, such as boulder reefs to coastal structures, might provide valuable habitat which may, to some extent, compensate for that lost and altered by urban infrastructure (e.g., Iverson and Bannerot, 1984). In Kogarah Bay, Sydney, Australia, small patches of salt marsh have actually been incorporated into the shoreline armoring at mid-tidal level to provide patches of this sparse habitat, although there have been no quantitative studies to date of the efficacy of such small-scale engineering (M.G. Chapman, personal observation). Active creation and restoration of dune and beach habitats on a large scale are major elements of the strategy adopted by the Netherlands to sustainably prepare for sea-level rise (e.g., Kabat et al., 2009).

Other attempts to reduce effects of past shoreline alterations and armoring come under the heading of managed retreat or coastal realignment (Townend and Pethick, 2002; Morris et al., 2004; French, 2008). Under this category, an approach recently developed to address coastal erosion for marshlands in Europe allows the sea to break through barriers and re-create fringing habitats, including salt marshes and lagoons (e.g., EC, 2004; Hughes and Paramor, 2004; Kabat et al., 2005). This involves dismantling or breaching walls, allowing the tidal flow to spread inland along natural topographic features (French, 2008; Townend, 2008). In Puget Sound (Washington, USA), case studies showed that the ecological responses to restoration of a beach (Olympic Sculpture Park, City of Seattle) via the removal of coastal armoring have been rapid and dramatic,

including the use of restored shallow-water habitats and benthic prey for foraging by three species of juvenile salmon (Toft et al., 2008). Although this approach has many positive attributes in that the shoreline is encouraged to find its own level, rather than be forced on to the environment, it has not been universally successful (French, 2008; Hughes et al., 2009) and is only appropriate where there is room for inland movement of the shore.

Where infrastructure is obligatory, it cannot be removed and therefore altering or building it to enhance its value as habitat, or to minimize its impact, must be the priority. Thus, coastal engineering that combines the expertise of ecologists and that of engineers is needed to evaluate new ways of building infrastructure. For example, a recent collaborative research program in Australia has evaluated adding novel intertidal habitats that mimic rock pools into featureless seawalls to increase diversity of species living on the wall itself (Chapman and Blockley, 2009). The design of these habitats was a compromise between the sorts of habitats that could be added with minimal cost and sustainable engineering standards and those that would retain water during low tide and thus potentially act as surrogate pools. Diversity of intertidal flora and fauna was increased manifold, by the simple addition of these structures.

Similarly, although there are plenty of data showing correlations between the extent of infrastructure and changes to diversity of a suite of taxa, research must start to go further to investigate which specific features of armoring affect diversity, especially where armored shores alter many features of habitat and environmental conditions at the same time. In addition, because biodiversity of some coastal habitats is maintained by interactions among species, more research on how armoring of shorelines changes basic interactions of among species, or among animals and their resources, is essential (Ivesa et al., 2010). Until we learn how much our ecological knowledge of natural habitats can be attributable to the novel ecosystems created in areas with excessive infrastructure and shoreline armoring, we will not be able to evaluate how much we actually know about these environments, or how much we need to consider that they may be such novel environments that current ecological understanding is not applicable (Hobbs et al., 2006).

8.02.8 Future of Shore Structures – Climate Change and Coastal Squeeze

Sea levels are predicted to rise over the twenty-first century due to a combination of factors associated with climatic change, including thermal expansion of the oceans, melting of the polar ice caps, changes to glacial ice masses, and uncertainty in terrestrial water storage (Church, 2001; Meehl et al., 2007). The average rate of sea-level rise is predicted to increase from 2 to 4 mm yr⁻¹ by 2100 (Meehl et al., 2007), equating to an average rise of between 59 cm (Meehl et al., 2007) and up to 200 cm, depending on the effects of glacial processes (Pfeffer et al., 2008). Coastal storms are also predicted to become more intense and perhaps more frequent (Webster et al., 2005; FitzGerald et al., 2008), increasing their destructive force by up to 25% (Scavia et al., 2002). Changes to sea levels and increased destructiveness of storms may cause accelerated erosion along sedimentary shorelines and consequently inland

retreat of coastlines (Slott et al., 2006). As shorelines retreat, society will be strongly pressed to protect infrastructure and citizens; in many cases, coastal armoring will be the most likely engineering solution to mitigate these threats (Polome et al., 2005). Thus, the consequence of greater threats to coastal assets and the response of societies to mitigate those threats will likely be that more coasts in more areas of the world will be armored. Predicted sea-level rise and increased storminess will not only intensify beach erosion and cause the use of coastal armoring to expand, but also effectively shift the location of many existing armoring structures to lower positions on the shore profile, thereby increasing the physical and ecological impacts of existing armoring to coastal ecosystems. Given the projected expansion of coastal armoring, juxtaposed against incomplete information on the ecological ramifications of these interventions, obtaining further evidence on the ecological effects of coastal armoring is a critical step toward environmentally sustainable coastal management.

8.02.9 A Way Forward

Although armoring of shorelines will continue to proliferate with increasing amounts of urbanization and the changes to sea level and weather patterns predicted by climatic change (Thompson et al., 2002), a coordinated approach to managing human populations on the coasts, similar to that developed for other urban environments (e.g., Pickett et al., 2001), could help minimize impacts on all coastal ecosystems, including rocky and soft-sediment habitats.

Coastlines are now more dynamic than ever because of changing storm patterns and sea-level rise, placing human and natural communities at greater risk. The costs of these coastal hazards are increasing as coastal development continues and natural buffers, such as beaches, dunes, wetlands, oyster reefs, corals, mangroves, and seagrasses, are lost (Airoldi and Beck, 2007). Many of the shoreline protection strategies being considered by coastal populations around the world involve hard engineering, including the building of sea walls and flood barriers. Such solutions are expensive, and, as we have documented in this chapter, carry significant, but often poorly documented and unacknowledged, costs in the form of impacts on natural ecosystems and the subsequent loss of ecosystem functions and services, including food security, water filtration, storm buffering, nutrient cycling, biodiversity, wildlife support, and recreational and natural value. By contrast, there is increasing evidence, in many circumstances, that natural ecosystems may offer solutions of comparable engineering efficiency (e.g., Sheppard et al., 2005), with considerable economic savings and with the maintenance of collateral ecosystem services and functions. One of the areas where there are real opportunities for identifying win-win solutions for human and natural communities is in building approaches that combine hazard mitigation and biodiversity conservation in coastal zones to preserve infrastructure, protect human communities, and preserve their livelihoods (Kareiva and Marvier, 2007). Hale et al. (2009) have recently shown the benefits of ecosystem-based adaptation in marine and coastal areas and recommended a number of guiding principles for developing effective ecosystem-based adaptation strategies. These and other innovative ecologically informed schemes

could provide coastal populations with viable and sustainable approaches to meet the formidable challenges and enjoy the benefits of life on the edge of the sea.

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References

- Able, K.W., Manderson, J.P., Studholme, A.L., 1998. The distribution of shallow water juvenile fishes in an urban estuary: the effects of manmade structures in the Lower Hudson River. *Estuaries* 21, 731–744.
- Able, K.W., Manderson, J.P., Studholme, A.L., 1999. Habitat quality for shallow water fishes in an urban estuary: the effects of man-made structures on growth. *Marine Ecology Progress Series* 187, 227–235.
- Adam, P., 2002. Saltmarshes in a time of change. *Environmental Conservation* 29 (1), 39–61.
- Airolidi, L., Abbiati, M., Beck, M.W., Hawkins, S.J., Jonsson, P.R., Martin, D., Moschella, P.S., Sundelöf, A., Thompson, R.C., Åberg, P., 2005a. An ecological perspective on the deployment and design of low-crested and other hard coastal defence structures. *Coastal Engineering* 52, 1073–1087.
- Airolidi, L., Bacchiocchi, F., Cagliola, C., Bulleri, F., Abbiati, M., 2005b. Impact of recreational harvesting on assemblages in artificial rocky habitats. *Marine Ecology Progress Series* 299, 55–66.
- Airolidi, L., Beck, M.W., 2007. Loss, status and trends for coastal marine habitats of Europe. *Oceanography and Marine Biology: An Annual Review* 45, 345–405.
- Allen, J.R.L., 2000. Morphodynamics of Holocene saltmarshes: a review sketch from the Atlantic and Southern North Sea coasts of Europe. *Quaternary Science Reviews* 19, 1155–1231.
- Arenas, F., Bishop, J.D.D., Carlton, J.T., Dyrinda, P.J., Farnham, W.F., Gonzalez, D.J., Jacobs, M.W., Lambert, C., Lambert, G., Nielsen, S.E., Pederson, J.A., Porter, J.S., Ward, S., Wood, C.A., 2006. Alien species and other notable records from a rapid assessment survey of marinas on the south coast of England. *Journal of the Marine Biological Association of the United Kingdom* 86 (6), 1329–1337.
- Attrill, M.J., Bilton, D.T., Rowden, A.A., Rundle, S.D., Thomas, R.M., 1999. The impact of encroachment and bankside development on the habitat complexity and supralittoral invertebrate communities of the Thames Estuary foreshore. *Aquatic Conservation: Marine and Freshwater Ecosystems* 9, 237–247.
- Bacchiocchi, F., Airolidi, L., 2003. Distribution and dynamics of epibiota on hard structures for coastal protection. *Estuarine Coastal and Shelf Science* 56, 1157–1166.
- Baine, M., 2001. Artificial reefs: a review of their design, application, management and performance. *Ocean and Coastal Management* 44, 241–259.
- Barros, F., 2001. Ghost crabs as a tool for rapid assessment of human impacts on exposed sandy beaches. *Biological Conservation* 97, 399–404.
- Bertasi, F., Colangelo, M.A., Abbiati, M., Ceccherelli, V.U., 2007. Effects of an artificial protection structure on the sandy shore macrofaunal community: the special case of Lido di Dante (Northern Adriatic Sea). *Hydrobiologia* 586, 277–290.
- Bilkovic, D.M., Roggero, M., Hershner, C.H., Havens, K.H., 2006. Influence of land use on macrobenthic communities in nearshore estuarine habitats. *Estuaries and Coasts* 29 (6b), 1185–1195.
- Bilkovic, D.M., Roggero, M.M., 2008. Effects of coastal development on nearshore estuarine nekton communities. *Marine Ecology Progress Series* 358, 27–39.
- Bird, B.L., Branch, L.C., Miller, D.L., 2004. Effects of coastal lighting on foraging behavior of beach mice. *Conservation Biology* 18, 1435–1439.
- Bird, E.C.F., 2000. Coastal Geomorphology: An Introduction. Wiley, Chichester, 322 pp.
- Bondesan, M., Castiglioni, G.B., Elmi, C., Gabbianelli, G., Marocco, R., Pirazzoli, P.A., Tomasin, A., 1995. Coastal areas at risk from storm surges and sea-level rise in Northeastern Italy. *Journal of Coastal Research* 11, 1354–1379.
- Bozek, C.M., Burdick, D.M., 2005. Impacts of seawalls on saltmarsh plant communities in the Great Bay Estuary, New Hampshire, USA. *Wetlands Ecology and Management* 13, 553–568.
- Brown, S.C., Hickey, C., Harrington, B., Gill, R., (Eds.), 2001. The U.S. Shorebird Conservation Plan, Second ed. Manomet Center for Conservation Sciences, Plymouth, MA, 61 pp.
- Bulleri, F., 2005a. The introduction of artificial structures on marine soft- and hard-bottoms: ecological implications of epibiota. *Environmental Conservation* 32, 101–102.
- Bulleri, F., 2005b. Role of recruitment in causing differences between intertidal assemblages on seawalls and rocky shores. *Marine Ecology Progress Series* 287, 53–64.
- Bulleri, F., 2006. Is it time for urban ecology to include the marine realm? *Trends in Ecology and Evolution* 21, 658–659.
- Bulleri, F., Airolidi, L., 2005. Artificial marine structures facilitate the spread of a non-indigenous green alga, *Codium fragile* ssp. *tomentosoides*, in the North Adriatic Sea. *Journal of Applied Ecology* 42, 1063–1072.
- Bulleri, F., Chapman, M.G., 2010. The introduction of coastal infrastructure as a driver of change in marine environments. *Journal of Applied Ecology* 47 (1), 26–35.
- Bulleri, F., Chapman, M.G., Underwood, A.J., 2005. Intertidal assemblages on seawalls and vertical rocky shores in Sydney Harbour, Australia. *Austral Ecology* 30, 655–667.
- Burger, J., 1994. The effect of human disturbance on foraging behavior and habitat use in piping plover (*Charadrius melodus*). *Estuaries* 17 (3), 695–701.
- Burke, L., Kura, Y., Kassem, K., Revenga, C., Spalding, M., McAllister, D., 2001. Coastal Ecosystems. World Resources Institute. http://pdf.wri.org/page_coastal.pdf (accessed March 2011).
- Burt, J., Bartholomew, A., Usseglio, P., Bauman, A., Sale, P.F., 2009. Are artificial reefs surrogates of natural habitats for corals and fish in Dubai, United Arab Emirates? *Coral Reefs* 28, 663–675.
- Bush, D.M., Pilkey, O.H., Neal, W.J., 2001. Coastal topography, human impact on. In: Steele, J.H., Thorpe, S.A., Turekian, K.K. (Eds.), *Encyclopedia of Ocean Sciences*. Academic Press, San Diego.
- Carlton, J.T., Hodder, J., 2003. Maritime mammals: terrestrial mammals as consumers in marine intertidal communities. *Marine Ecology Progress Series* 256, 271–286.
- Carter, J.W., Carpenter, A.L., Foster, M.S., Jessee, W.N., 1985. Benthic succession on an artificial reef designed to support a kelp-reef community. *Bulletin of Marine Science* 37, 86–113.
- Cencini, C., 1998. Physical processes and human activities in the evolution of the Po delta, Italy. *Journal of Coastal Research* 14, 774–793.
- Chapman, M.G., 2003. Paucity of mobile species on constructed seawalls: effects of urbanization on biodiversity. *Marine Ecology Progress Series* 264, 21–29.
- Chapman, M.G., Bulleri, F., 2003. Intertidal seawalls – new features of landscape in intertidal environments. *Landscape and Urban Planning* 62, 159–172.
- Chapman, M.G., Blockley, D.G., 2009. Engineering novel habitats on urban infrastructure to increase intertidal biodiversity. *Oecologia* 161, 625–635.
- Chapman, M.G., Clynick, B.G., 2006. Experiments testing the use of waste material in estuaries as habitat for subtidal organisms. *Journal of Experimental Marine Biology and Ecology* 338, 164–178.
- Chapman, M.G., Roberts, D.E., 2004. Use of seagrass wrack in restoring disturbed Australian saltmarshes. *Ecological Management and Restoration* 5, 183–190.
- Chapman, M.G., Underwood, A.J., 2009. Comparative effects of urbanization in marine and terrestrial habitats. In: McDonnell, M.J., Haas, A.K., Breuste, J.H. (Eds.), *Ecology of Cities and Towns: A Comparative Approach*. Cambridge University Press, New York, NY, pp. 51–70.
- Charlier, R.H., Chaineux, M.C.P., Morcos, S., 2005. Panorama of the history of coastal protection. *Journal of Coastal Research* 21 (1), 79–111.
- Christoffers, E.W., 1986. Ecology of the ghost crab *Ocypode quadrata* (Fabricius) on Assateague Island, Maryland and the impacts of various human uses of the beach on

- their distribution and abundance. Ph.D. Thesis, Michigan State University, Ann Harbor, MI, USA.
- Church, J.A., 2001. How fast are sea levels rising? *Science* 294, 802–803.
- Clark, J.R., 1996. Coastal Zone Management Handbook. CRC Press, Boca Raton, FL, 694 pp.
- Clynick, B.G., 2006. Assemblages of fish associated with coastal marinas in north-western Italy. *Journal of the Marine Biological Association of the United Kingdom* 86 (4), 847–852.
- Clynick, B.G., 2008. Characteristics of an urban fish assemblage: distribution of fish associated with coastal marinas. *Marine Environmental Research* 65 (1), 18–33.
- Clynick, B.G., Chapman, M.G., Underwood, A.J., 2007. Effects of epibiota on assemblages of fish associated with urban structures. *Marine Ecology Progress Series* 332, 201–210.
- Colombini, I., Chelazzi, L., 2003. Influence of marine allochthonous input on sandy beach communities. *Oceanography and Marine Biology: Annual Review* 41, 115–159.
- Colosio, F., Abbiati, M., Airolidi, L., 2007. Effects of beach nourishment on sediments and benthic assemblages. *Marine Pollution Bulletin* 54 (8), 1197–1206.
- Connell, S.D., 2000. Floating pontoons create novel habitats for subtidal epibiota. *Journal of Experimental Marine Biology and Ecology* 247, 183–194.
- Connell, S.D., 2001. Predatory fish do not always affect the early development of epibenthic assemblages. *Journal of Experimental Marine Biology and Ecology* 260, 1–12.
- Costanza, R., d'Arge, R., deGroot, R., Farber, S., Grasso, M., Hannon, B., Limburg, K., Naeem, S., O'Neill, R.V., Paruelo, J., Raskin, R.G., Sutton, P., van den Belt, M., 1997. The value of the world's ecosystem services and natural capital. *Nature* 387 (6630), 253–260.
- Dafforn, K.A., Johnston, E.L., Glasby, T.M., 2009. Shallow moving structures promote marine invader dominance. *Biofouling* 25, 277–287.
- Davis, J.L.D., Levin, L.A., Walther, S.M., 2002. Artificial armored shorelines: sites for open-coast species in a southern California Bay. *Marine Biology* 140, 1249–1262.
- Defeo, O., McLachlan, A., Schoeman, D., Schlacher, T., Dugan, J., Jones, A., Lastra, M., Scapini, F., 2009. Threats to sandy beach ecosystems: a review. *Estuarine, Coastal and Shelf Science* 81, 1–12.
- DeLuca, W.V., Studds, C.E., King, R.S., Marra, P.P., 2008. Coastal urbanization and the integrity of estuarine waterbird communities: threshold responses and the importance of scale. *Biological Conservation* 141 (11), 2669–2678.
- Dethier, M.N., McDonald, K., Strathmann, R.R., 2003. Colonization and connectivity of habitat patches for coastal marine species distant from source populations. *Conservation Biology* 17, 1024–1035.
- Dong, P., 2004. An assessment of groyne performance in the United Kingdom. *Coastal Management* 32, 203–213.
- Doody, J.P., 2004. Coastal squeeze: a historical perspective. *Journal of Coastal Conservation* 10, 138.
- Douglass, S., Pickel, B., 1999. The tide doesn't go out anymore – the effect of bulkheads on urban bay shorelines. *Shore and Beach* 67, 19–25.
- Duarte, C.M., 2002. The future of seagrass meadows. *Environmental Conservation* 29, 192–206.
- Dugan, J.E., Hubbard, D.M., 2006. Ecological responses to coastal armoring on exposed sandy beaches. *Shore and Beach* 74 (1), 10–16.
- Dugan, J.E., Hubbard, D.M., 2010. Loss of coastal strand habitat in southern California: the role of beach grooming. *Estuaries and Coasts* 33 (1), 67–77.
- Dugan, J.E., Hubbard, D.M., McCrary, M., Pierson, M., 2003. The response of macrofauna communities and shorebirds to macrophyte wrack subsidies on exposed beaches of southern California. *Estuarine, Coastal and Shelf Science* 58S, 133–148.
- Dugan, J.E., Hubbard, D.M., Page, H.M., Schimel, J., 2011. Marine macrophyte wrack inputs and dissolved nutrients in beach sands. *Estuaries and Coasts* DOI: 10.1007/s12237-011-3975-9.
- Dugan, J.E., Hubbard, D.M., Rodil, I.F., Revell, D., 2008. Ecological effects of coastal armoring on sandy beaches. *Marine Ecology* 29, 160–170.
- EC, 2004. Living with Coastal Erosion in Europe – Sediment and Space for Sustainability. OPOCE, Luxembourg. http://www.eurosion.org/project/eurosion_en.pdf (accessed March 2011).
- EEA, 1999. State and Pressures of the Marine and Coastal Mediterranean Environment. Environmental Issues Series 5. Luxembourg: OPOCE. Online. <http://reports.eea.europa.eu/ENVSERIES05/en/envissue05.pdf> (accessed 15 April 2011).
- Fanini, L., Marchetti, G.M., Scapini, F., Defeo, O., 2009. Effects of beach nourishment and groynes building on population and community descriptors of mobile arthropodofauna. *Ecological Indicators* 9, 167–178.
- Fauvelot, C., Bertozzi, F., Costantini, F., Airolidi, L., Abbiati, M., 2009. Lower genetic diversity in the limpet *Patella caerulea* on urban coastal structures compared to natural rocky habitats. *Marine Biology* 156, 2313–2323.
- Feagin, R.A., Sherman, D.J., Grant, W.E., 2005. Coastal erosion, global sea-level rise, and the loss of sand dune plant habitats. *Frontiers in Ecology and the Environment* 7 (3), 359–364.
- Fitzgerald, D.M., Fenster, M.S., Argow, B.A., Buynevich, I.V., 2008. Coastal impacts due to sea-level rise. *Annual Review of Earth and Planetary Sciences* 36, 601–647.
- Fletcher, C.H., Mullane, R.A., Richmond, B.M., 1997. Beach loss along armored shorelines on Oahu, Hawaiian Islands. *Journal of Coastal Research* 13, 209–215.
- Florida DEP, Department of Environmental Protection Bureau of Beaches and Coastal Systems, 1990. Coastal Armoring in Florida, Final Status Report, December 1990, Tallahassee, FL.
- French, J.R., 2008. Hydrodynamic modelling of estuarine flood defence realignment as an adaptive management response to sea-level rise. *Journal of Coastal Research: Issue 2 supplement*, 1–12.
- French, P.W., 1997. Coastal and Estuarine Management. Routledge, London, 251 pp.
- Glasby, T.M., 1999. Differences between subtidal epibiota on pier pilings and rocky reefs at marinas in Sydney. *Estuarine, Coastal and Shelf Science* 48, 281–290.
- Glasby, T.M., Connell, S.D., 1999. Urban structures as marine habitats. *Ambio* 28, 595–598.
- Glasby, T.M., Connell, S.D., Holloway, M.G., Hewitt, C.L., 2007. Nonindigenous biota on artificial structures: could habitat creation facilitate biological invasions? *Marine Biology* 151, 887–895.
- Goodsell, P.J., Chapman, M.G., Underwood, A.J., 2007. Differences between biota in anthropogenically fragmented habitats and in naturally patchy habitats. *Marine Ecology Progress Series* 351, 15–23.
- Griggs, G.B., 1998. The armoring of California's coast. In: Magoon, O.T., Converse, H., Baird, B., Miller-Henson, M. (Eds.), *California and the World Ocean '97, Conference Proceedings*. American Society of Civil Engineers, Reston, VA, pp. 515–526.
- Griggs, G.B., 1999. The protection of California's coast: past, present and future. *Shore and Beach* 67 (1), 18–28.
- Griggs, G.B., 2005a. California's retreating coastline: where do we go from here? In: Santa Barbara, C.A., Magoon, O.T., Converse, H., Baird, B., Miller-Henson, M. (Eds.), *California and the World Ocean, Conference Proceedings*, October 2002. American Society of Civil Engineers, Reston, VA, pp. 121–125.
- Griggs, G.B., 2005b. The Impacts of Coastal Armoring. *Shore and Beach* 73 (1), 13–22.
- Hale, L.Z., Meliane, I., Davidson, S., Sandwith, T., Beck, M., Hoekstra, J., Spalding, M., Murawski, S., Cyr, N., Osgood, K., Hatzios, M., Van Eijk, P., Davidson, N., Eichbaum, W., Dreus, C., Obura, D., Tamelander, J., Herr, D., McClennen, C., Marshall, P., 2009. Ecosystem-based adaptation in marine and coastal ecosystems. *Renewable Resources Journal* 25, 21–28.
- Hall, M.J., Pilkey, O.H., 1991. Effects of hard stabilization on dry beach widths for New Jersey. *Journal of Coastal Research* 7 (3), 771–785.
- Hansen, B., 2007. Weathering the storm: the Galveston seawall and grade raising. *Civil Engineering* 77 (4), 32–33.
- Harangozo, S.A., 1992. Flooding in the Maldives and its implications for the global sea level rise debate. In: Woodworth P.L., Pugh, D.T., de Ronde, J.G., Warrick, R.G., Hannah, J. (Eds.), *Sea Level Changes: Determination and Effects*. American Geophysical Union, Washington, DC, IUGG, vol. 11, *Geophysical Monograph* 69.
- Harmsworth, G.C., Long, S.P., 1986. An assessment of saltmarsh erosion in Essex, England, with reference to the Dengie Peninsula. *Biological Conservation* 35, 377–387.
- Hobbs, R.J., Arico, S., Aronson, J., Baron, J.S., Bridgewater, P., Cramer, V.A., Epstein, P.R., Ewel, J.J., Klink, C.A., Lugo, A.E., Norton, D., Ojima, D., Richardson, D.M., Sanderson, E.W., Valladares, F., Vilá, M., Zamora, R., Zobel, M., 2006. Novel ecosystems: theoretical and management aspects of the new ecological world order. *Global Ecology and Biogeography* 15, 1–7.
- Hoegh-Guldberg, O., Hughes, L., McIntyre, S., Lindenmayer, D.B., Parmesan, C., Possingham, H.P., Thomas, C.D., 2008. Assisted colonization and rapid climate change. *Science* 321, 345–346.
- Howe, M.A., Geissler, P.H., Harrington, B.A., 1989. Population trends of North American shorebirds based on the International Shorebird Survey. *Biological Conservation* 49, 185–200.
- Hubbard, D.M., Dugan, J.E., 2003. Shorebird use of an exposed sandy beach in southern California. *Estuarine, Coastal and Shelf Science* 58S, 169–182.
- Hughes, R.G., Fletcher, P.W., Hardy, M.J., 2009. Successional development of saltmarsh in two managed realignment areas in SE England, and prospects for saltmarsh restoration. *Marine Ecology Progress Series* 384, 13–22.
- Hughes, R.G., Paramor, A.L., 2004. On the loss of saltmarshes in south-east England and methods for their restoration. *Journal of Applied Ecology* 41, 440–448.
- Iannuzzi, T.J., Weinstein, M.P., Sellner, K.G., Barrett, J.C., 1996. Habitat disturbance and marina development: an assessment of ecological effects.1. Changes in primary production due to dredging and marina construction. *Estuaries* 19, 257–271.
- Inger, R., Attrill, M.J., Bearhop, S., Broderick, A.C., Grecian, W.J., Hodgson, D.J., Mills, C., Sheehan, E., Votier, S.C., Witt, M.J., Godley, B.J., 2009. Marine renewable

- energy: potential benefits to biodiversity? An urgent call for research. *Journal of Applied Ecology* 46 (6), 1145–1153.
- Iverson, E.S., Bannerot, S.P., 1984. Artificial reefs under marine docks in southern Florida. *North American Journal of Fisheries Management* 4, 294–199.
- Iveša, L., Chapman, M.G., Underwood, A.J., Murphy, R.J., 2010. Differential patterns of distribution of limpets on intertidal seawalls: experimental investigation of the roles of recruitment, survival and competition. *Marine Ecology Progress Series* 407, 55–69.
- Jackson, N.L., Smith, D.R., Nordstrom, K.F., 2008. Physical and chemical changes in the foreshore of an estuarine beach: implications for viability and development of horseshoe crab *Limulus polyphemus* eggs. *Marine Ecology Progress Series* 355, 209–218.
- Jaramillo, E., Contreras, H., Bollinger, A., 2002b. Beach and faunal response to the construction of a seawall in a sandy beach of south central Chile. *Journal of Coastal Research* 18 (3), 523–529.
- Jaramillo, E., Dugan, J., Contreras, H., 2002a. Abundance, tidal movement, population structure and burrowing rate of *Emerita analoga* (Stimpson 1857) (Anomura, Hippidae) at a dissipative and a reflective beach in south central Chile. *Marine Ecology Napoli* 21 (2), 113–127.
- Jeffic, L., Bernhard, M., Demetropoulos, A., Fernex, F., Gabrielides, G.P., Gasparovic, F., Halim, Y., Orhon, D., & Saliba, L.J., 1990. State of the Marine Environment in the Mediterranean Region. UNEP Regional Seas Reports and Studies 132/1990 and MAP Technical Reports Series 28/1989. Athens UNEP. Online. <http://195.97.36.231/acrobathiles/>
- Johannesson, K., Warmoes, T., 1990. Rapid colonization of Belgian breakwaters by the direct developer, *Littorina-saxatilis* (Oliv) (Prosobranchia, Mollusca). *Hydrobiologia* 193, 99–108.
- John, D.M., Pople, W., 1973. The fish grazing of rocky shore algae in the Gulf of Guinea. *Journal of Experimental Marine Biology and Ecology* 11, 81–90.
- Jones, G., 1994. Global warming, sea level change and the impact on estuaries. *Marine Pollution Bulletin* 28, 7–14.
- Kabat, P., Fresco, L.O., Stive, M.J.F., Veerman, C.P., van Alphen, J.S.L.J., Parmet, B.W.A.H., Hazeleger, W., Katsman, C.A., 2009. Dutch coasts in transition. *Nature Geoscience* 2 (7), 450–452.
- Kabat, P., van Vierssen, W., Veraart, J., Vellinga, P., Aerts, J., 2005. Climate proofing the Netherlands. *Nature* 438 (7066), 283–284.
- Kareiva, P., Marvier, M., 2007. Conversation for the people – pitting nature and biodiversity against people makes little sense. Many conservationists now argue that human health and well-being should be central to conservation efforts. *Scientific American* 297 (4), 50–57.
- Kendall, M.A., Burrows, M.T., Southward, A.J., Hawkins, S.J., 2004. Predicting the effects of marine climate change on the invertebrate prey of the birds of rocky shores. *Ibis* 146 (S1), 40–47.
- Kennish, M.J., 2002. Environmental threats and environmental future of estuaries. *Environmental Conservation* 29, 78–107.
- Kersten, M., Piersma, T., 1987. High levels of energy expenditure in shorebirds: metabolic adaptations to an energetically expensive way of life. *Ardea* 75, 175–187.
- King, R.S., Deluca, W.V., Whigham, D.F., Marra, P.P., 2007. Threshold effects of coastal urbanization on *Phragmites australis* (common reed) abundance and foliar nitrogen in Chesapeake Bay. *Estuaries and Coasts* 30 (3), 469–481.
- Klapow, L.A., 1972. Fortnightly molting and reproductive cycles in the sand-beach isopod, *Excirolana chiltoni*. *Biological Bulletin* 143, 568–591.
- Klein, J.C., Underwood, A.J., Chapman, M.G., 2011. Urban structures provide new insights into interactions among grazers and habitat. *Ecological Applications* 21, 427–438.
- Koike, K., 1993. The countermeasures against coastal hazards in Japan. *Geojournal* 38, 301–312.
- Kraus, N.C., McDougal, W.G., 1996. The effects of seawalls on the beach: part 1, an updated literature review. *Journal of Coastal Research* 12 (3), 691–701.
- Lathrop, R.G., Jr., Love, A., 2007. Vulnerability of New Jersey's coastal habitats to sea level rise. Unpublished report. Grant F. Walton Center for Remote Sensing & Spatial Analysis Rutgers University and the American Littoral Society Highlands, NJ.
- Laurie, M., 1979. Nature and city planning in the nineteenth century. In: Laurie, I.C. (Ed.), *Nature in Cities*. Wiley, Chichester, pp. 37–63.
- Lee, M., 2001. Coastal defence and the Habitats Directive: predictions of habitat change in England and Wales. *The Geographical Journal* 167 (1), 39–56.
- Levings, C.D., 1991. Intertidal habitats used by juvenile chinook salmon (*Oncorhynchus tshawytscha*) rearing in the north arm of the Fraser River Estuary. *Marine Pollution Bulletin* 22 (1), 20–26.
- Living Shoreline Summit Steering Committee, 2006. Preface. Proceedings of the 2006 Living Shoreline Summit. CRC Publ. No. 08-164, Chesapeake Bay.
- Locke, A., Hanson, J.M., Ellis, K.M., Thompson, J., Rochette, R., 2007. Invasion of the southern Gulf of St. Lawrence by the clubbed tunicate (*Styela clava* Herdman): potential mechanisms for invasions of Prince Edward Island estuaries. *Journal of Experimental Marine Biology and Ecology* 342 (1) 69–77.
- Long, B.G., Dennis, D.M., Skewes, T.D., Poiner, I.R., 1996. Detecting an environmental impact of dredging on seagrass beds with a BACIR sampling design. *Aquatic Botany* 53, 235–243.
- Lotze, H.K., Reise, K., Worm, B., Van Beusekom, J., Busch, M., Ehlers, A., Heinrich, D., Hoffmann, R.C., Holm, P., Jensen, C., Knottnerus, O.S., Langhanki, N., Prummel, W., Vollmer, M., Wolff, W.J., 2005. Human transformations of the Wadden Sea ecosystem through time: a synthesis. *Helgolander Marine Research* 59, 84–95.
- Lucrezi, S., Schlacher, T.A., Walker, S.J., 2009. Monitoring human impacts on sandy shore ecosystems: a test of ghost crabs (*Ocypode* spp.) as biological indicators on an urban beach. *Environmental Monitoring and Assessment* 152, 413–424.
- Mann, R.B., 1988. Ten trends in the continuing renaissance of urban waterfronts. *Landscape and Urban Planning* 16, 177–199.
- Martin, D., Bertasi, F., Colangelo, M.A., de Vries, M., Frost, M., Hawkins, S.J., Macpherson, E., Moschella, P.S., Satta, M.P., Thompson, R.C., Ceccherelli, V.U., 2005. Ecological impact of coastal defence structures on sediments and mobile infauna: evaluating and forecasting consequences of unavoidable modifications of native habitats. *Coastal Engineering* 52, 1027–1051.
- Martins, G.M., Amaral, A.F., Wallenstein, F.M., Neto, A.I., 2009. Influence of a breakwater on nearby rocky intertidal community structure. *Marine Environmental Research* 67, 237–245.
- McLachlan, A., Jaramillo, E., 1995. Zonation on sandy shores. *Oceanography and Marine Biology: An Annual Review* 33, 305–335.
- McLachlan, A., Wooldridge, T., Van der Horst, G., 1979. Tidal movements of the macrofauna on an exposed sandy beach in South Africa. *Journal of Zoology (London)* 188, 433–442.
- Meehl, G.A., Stocker, T.F., Collins, W.D., Friedlingstein, P., Gaye, A.T., Gregory, J.M., Kitoh, A., Knutti, R., Murphy, J.M., Noda, A., Raper, S.C.B., Watterson, I.G., Weaver, A.J., Zhao, Z.-C., 2007. Global climate projections. In: Solomon, S., Qin, D., Manning, M., Chen, Z., Marquis, M., Averyt, K.B., Tignor, M., Miller, H.L. (Eds.) *Climate Change 2007. The Physical Science Basis. Contribution of Working Group 1 to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change*. Cambridge University Press, Cambridge, pp. 749–844.
- Meinesz, A., Lefevre, J.R., Astier, J.M., 1991. Impact of coastal development on the infralittoral zone along the southeastern Mediterranean shore of continental France. *Marine Pollution Bulletin* 23, 343–347.
- Miles, J.R., Russell, P.E., Huntley, D.A., 2001. Field measurements of sediment dynamics in front of a seawall. *Journal of Coastal Research* 17 (1), 195–206.
- Miller, M.W., Valdivia, A., Kramer, K.L., Mason, B., Williams, D.E., Johnston, L., 2009. Alternate benthic assemblages on reef restoration structures and cascading effects on coral settlement. *Marine Ecology Progress Series* 387, 147–156.
- Mimura, N., Nunn, P.D., 1998. Trends of beach erosion and shoreline protection in rural Fiji. *Journal of Coastal Research* 14 (1), 37–46.
- Moiser, A.E., Witherington, B.E., 2002. Documented effects of coastal armoring structures on sea turtle nesting behavior. In: Mosier A., Foley A., Brost B. (Eds.), *Proceedings of the Twentieth Annual Symposium on Sea Turtle Biology and Conservation*. NOAA Tech. Memo. NMFS-SEFSC-477.
- Moreira, J., 2006. Patterns of occurrence of grazing molluscs on sandstone and concrete seawalls in Sydney Harbour (Australia). *Molluscan Research* 26, 51–60.
- Moreira, J., Chapman, M.G., Underwood, A.J., 2006. Seawalls do not sustain viable populations of limpets. *Marine Ecology Progress Series* 322, 179–188.
- Morris, J.T., Sundareshwar, P.V., Nietch, C.T., Kjerfve, B., Cahoon, D.R., 2002. Responses of coastal wetlands to rising sea level. *Ecology* 83 (10), 2869–2877.
- Morris, R.K.A., Reach, I.S., Duffy, M.J., Collins, T.S., Leafe, R.N., 2004. On the loss of saltmarshes in south-east England and the relationship with *Nereis diversicolor*. *Journal of Applied Ecology* 41, 787–791.
- Morton, R.B., 1992. Fish assemblages in residential canal developments near the mouth of a subtropical Queensland estuary. *Australian Journal of Marine and Freshwater Research* 43, 1359–1371.
- Moschella, P.S., Abbiati, M., Lberg, P., Airolidi, L., Anderson, J.M., Bacchiocchi, F., Bulleri, F., Dunesen, G.E., Frost, M., Gacia, E., Grannhag, L., Jonsson, P.R., Satta, M.P., Sundelof, A., Thompson, R.C., Hawkins, S.J., 2005. Low-crested coastal defence structures as artificial habitats for marine life: using ecological criteria in design. *Coastal Engineering* 52, 1053–1071.
- Nakashima, B.S., Taggart, C.T., 2002. Is beach-spawning success for capelin, *Mallotus villosus* (Müller), a function of the beach? *ICES Journal of Marine Science* 59, 897–908.
- Neill, P.E., Alcalde, O., Faugeton, S., Navarrete S.A., Correa J.A., 2006. Invasion of *Codium fragile* ssp. *tomentosoides* in northern Chile: a new threat for *Gracilaria* farming. *Aquaculture* 259, 202–210.
- Nicholls, R.J., Hoozemans, F.M.J., Marchand, M., 1999. Increasing flood risk and wetland losses due to global sea-level rise: regional and global analyses. *Global Environmental Change – Human and Policy Dimensions* 9, S69–S87.

- Niles, L.J., Bart, J., Sitters, H.P., Dey, A.D., Clark, K.E., Atkinson, P.W., Baker, A.J., Bennett, K.A., Kalasz, K.S., Clark, N.A., Clark, J., Gillings, S., Gates, A.S., Gonzalez, P.M., Hernandez, D.E., Minton, C.D.T., Morrison, R.I.G., Porter, R.R., Ross, R.K., Veitch, C.R., 2009. Effects of horseshoe crab harvest in Delaware Bay on red knots: are harvest restrictions working? *BioScience* 59 (2), 153–164.
- Nordstrom, K.F., 2000. *Beaches and Dunes of Developed Coasts*. Cambridge University Press, Cambridge.
- NRC, 2007. *Mitigating Shoreline Erosion along Sheltered Coasts*. Ocean Study Board, National Research Council. National Academies Press, Washington, DC.
- Page, H.M., Culver, C., Dugan, J., Mardian, B., 2008. Oceanographic gradients and patterns in invertebrate assemblages on offshore oil platforms. *ICES Journal of Marine Science* 65, 851–861.
- Page, H.M., Dugan, J.E., Culver, C.C., Hoesterey, J., 2006. Exotic invertebrate species on offshore oil platforms. *Marine Ecology Progress Series* 325, 101–107.
- Partyka, M., Peterson, M.S., 2008. Habitat quality and salt-marsh species assemblages along an anthropogenic estuarine landscape. *Journal of Coastal Research* 24 (6), 1570–1581.
- Patsch, K., Griggs, G., 2008. A sand budget for the Santa Barbara Littoral Cell, California. *Marine Geology* 252, 50–61.
- Pattiaratchi, C.B., Olsson, D., Hetzel, Y., Lowe, R., 2009. Wave-driven circulation patterns in the lee of groynes. *Continental Shelf Research* 29 (16), 1961–1974.
- Pérez-Ruzafa, A., Garcia-Charton, J.A., Barcala, E., Marcos, C., 2006. Changes in benthic fish assemblages as a consequence of coastal works in a coastal lagoon: The Mar Menor (Spain, Western Mediterranean). *Marine Pollution Bulletin* 53, 107–120.
- Perkol-Finkel, S., Benayahu, Y., 2009. The role of differential survival patterns in shaping coral communities on neighboring artificial and natural reefs. *Journal of Experimental Marine Biology and Ecology* 369, 1–7.
- Perkol-Finkel, S., Shashar, N., Benayahu, Y., 2006. Can artificial reefs mimic natural reef communities? The roles of structural features and age. *Marine Environmental Research* 61, 121–135.
- Peterson, M.S., Comyns, B.H., Hendon, J.R., Bond, P.J., Duff, G.A., 2000. Habitat use by early life-history stages of fishes and crustaceans along a changing estuarine landscape: differences between natural and altered shoreline sites. *Wetlands Ecology and Management* 8, 209–219.
- Peterson, M.S., Lowe, M.R., 2009. Implications of cumulative impacts to estuarine and marine habitat quality for fish and invertebrate resources. *Reviews in Fisheries Science* 17 (4), 505–523.
- Pfeffer, W.T., Harper, J.T., O'Neil, S., 2008. Kinematic constraints on glacier contributions to 21st-century sea-level rise. *Science* 321, 1340–1343.
- Pickett, S.T.A., Cadenasso, M.L., Grove, J.M., Nilon, C.H., Pouyat, R.V., Zipperer, W.C., Costanza, R., 2001. Urban ecological systems: linking terrestrial ecological, physical and socioeconomic components of metropolitan areas. *Annual Review of Ecology and Systematics* 32, 127–157.
- Pinn, E.H., Mitchell, K., Corkill, J., 2005. The assemblages of groynes in relation to substratum age, aspect and microhabitat. *Estuarine, Coastal and Shelf Science* 62, 271–282.
- Pister, B., 2009. Urban marine ecology in southern California: the ability of riprap structures to serve as rocky intertidal habitat. *Marine Biology* 156, 861–873.
- Polis, G.A., Hurd, S.D., 1996. Linking marine and terrestrial food webs: allochthonous input from the ocean supports high secondary productivity on small island and coastal land communities. *American Naturalist* 147, 396–423.
- Polome, P., Marzetti, S., Van der Veen, A., 2005. Economic and social demands for coastal protection. *Coastal Engineering* 52, 819–840.
- Reeves, B., Bookheim, B., Berry, H., 2003. Using ShoreZone inventory data to identify potential forage fish spawning habitat. Abstract from the 2003 Georgia Basin/Puget Sound Research Conference.
- Reise, K., 2005. Coast of change: habitat loss and transformations in the Wadden Sea. *Helgolander Marine Research* 59, 9–21.
- Revell, D.L., Dugan, J.E., Hubbard, D.M., 2011. Physical and ecological responses to the 1997–98 El Niño. *Journal of Coastal Research* 24 (7).
- Rice, C.A., 2006. Effects of shoreline modification on a northern Puget Sound beach: microclimate and embryo mortality in surf smelt (*Hypomesus pretiosus*). *Estuaries and Coasts* 29 (1), 63–71.
- Rilov, G., Benayahu, Y., 1998. Vertical artificial structures as an alternative habitat for coral reef fishes in disturbed environments. *Marine Environmental Research* 45, 431–451.
- Rippon, S., 2000. *The Transformation of Coastal Wetlands: Exploitation and Management of Marshland Landscapes in North West Europe during the Roman and Medieval Periods*. British Academy, London.
- Rizkalla, C.E., Savage, A., 2011. Impact of seawalls on loggerhead sea turtle (*Caretta caretta*) nesting and hatching success. *Journal of Coastal Research* 27 (1), 166–173.
- Roberts, L., 1993. Wetland's trading is a loser's game, say ecologists. *Science* 260, 1890–1892.
- Ruiz, G.M., Carlton, J.T., Grosholz, E.D., Hines, A.H., 1997. Global invasions of marine and estuarine habitats by non-indigenous species: mechanisms, extent and consequences. *American Zoologist* 37, 621–632.
- Runyan, K., Griggs, G., 2003. The effects of armoring seaciffs on the natural sand supply to the beaches of California. *Journal of Coastal Research* 19 (2), 336–347.
- Russell, G., 2000. The algal vegetation of coastal defences: a case study from NW England. *Botanical Journal of Scotland* 52, 31–42.
- Sammarco, P.W., Atchison, A.D., Boland, G.S., 2004. Expansion of coral communities within the Northern Gulf of Mexico via offshore oil and gas platforms. *Marine Ecology Progress Series* 280, 129–143.
- Sax, D.F., Gaines, S.D., 2003. Species diversity: from global decreases to local increases. *Trends in Ecology and Evolution* 18 (11), 561–566.
- Scavia, D., Field, J.C., Boesch, D.F., Buddemeier, R.W., Burkett, V., Cayan, D.R., Fogarty, M., Harwell, M.A., Howarth, R.W., Mason, C., Reed, D.J., Royer, T.C., Sallenger, A.H., Titus, J.G., 2002. Climate change impacts on US coastal and marine ecosystems. *Estuaries* 25, 149–164.
- Schlacher, T.A., Dugan, J.E., Schoeman, D.S., Lastra, M., Jones, A., Scapini, F., McLachlan, A., Defeo, O., 2007. Sandy beaches at the brink. *Diversity and Distributions* 13 (5), 556–560.
- Seitz, R.D., Lipcius, R.N., Olmstead, N.H., Seebo, M.S., Lambert, D.M., 2006. Influence of shallow-water habitats and shoreline development on abundance, biomass and diversity of benthic prey and predators in Chesapeake Bay. *Marine Ecology Progress Series* 326, 11–27.
- Sheppard, C., Dixon, D.J., Gourlay, M., Sheppard, A., Payet R., 2005. Coral mortality increases wave energy reaching shores protected by reef flats: examples from the Seychelles. *Estuarine Coastal and Shelf Science* 64, 223–234.
- Sherman, D.J., Bauer, B.O., Nordstrom, K.F., Allen, J.R., 1990. A tracer study of sediment transport in the vicinity of a groin: New York, USA. *Journal of Coastal Research* 6 (2), 427–438.
- Short, F.T., Burdick, D.M., 1996. Quantifying eelgrass habitat loss in relation to housing development and nitrogen loading in Waquoit Bay, Massachusetts. *Estuaries* 19, 730–739.
- Slott, J.M., Murray, A.B., Ashton, A.D., Crowley, T.J., 2006. Coastline responses to changing storm patterns. *Geophysical Research Letters* 33, L18404. doi: 10.1029/2006GL027445.
- Smyder, E.A., Martin, K.L.M., 2002. Temperature effects on egg survival and hatching during extended incubation period of California grunion, *Leuresthes tenuis*. *Copeia* 2002, 313–320.
- Sobocinski, K.L., Cordell, J.R., Simenstad, C.A., 2010. Effects of shoreline modifications on supratidal macroinvertebrate fauna on Puget Sound, Washington beaches. *Estuaries and Coasts* 33, 699–711. doi:10.1007/s12237-009-9262-9.
- Suchanek, T.H., 1994. Temperate coastal marine communities: biodiversity and threats. *American Zoologist* 34, 110–114.
- Surfrider Foundation, 2010. State of the Beach Report. State Reports. <http://www.surfrider.org/stateofthebeach/05-sr/> (accessed March 2011).
- Svane, I., Petersen, J.K., 2001. On the problems of epibioses, fouling and artificial reefs, a review. *P.S.Z.N.I: Marine Ecology* 22, 169–188.
- Sweka, J.A., Smith, D.R., Millard, M.J., 2007. An age-structured population model for horseshoe crabs in the Delaware Bay area to assess harvest and egg availability for shorebirds. *Estuaries and Coasts* 30, 277–286.
- Terlizzi, A., Bevilacqua, S., Scuderi, D., Fiorentino, D., Guarnieri, G., Giangrande, A., Licciano, M., Felling, S., Frascetti, S., 2008. Effects of offshore platforms on soft-bottom macro-benthic assemblages: a case study in a Mediterranean gas field. *Marine Pollution Bulletin* 56 (7), 1303–1309.
- Thomalla, F., Vincent, C.E., 2003. Beach response to shore-parallel breakwaters at Sea Palling, Norfolk, UK. *Estuarine, Coastal and Shelf Science* 56, 203–212.
- Thompson, R.C., Crowe, T.P., Hawkins, S.J., 2002. Rocky intertidal communities: past environmental changes, present status and predictions for the next 25 years. *Environmental Conservation* 29, 168–191.
- Toft, J., Cordell, J., Heerhartz, S., Armbrust, E., Ogston, A., and Flemer, E., 2008. Olympic Sculpture Park—results from year 1 post-construction monitoring of shoreline habitats: Seattle, Wash., University of Washington, School of Aquatic and Fishery Sciences and Seattle Public Utilities, City of Seattle, Technical Report SAFS-UW-0801, 113 p.
- Toft, J.D., Cordell, J.R., Simenstad, C.A., Stamatou, L.A., 2007. Fish distribution, abundance, and behavior along city shoreline types in Puget Sound. *North American Journal of Fisheries Management* 27, 465–480.
- Townend, I., Pethick, J., 2002. Estuarine flooding and managed retreat. *Philosophical Transactions Royal Society of London A* 360, 1477–1495.
- Townend, I.H., 2008. Breach design for managed realignment sites. *Proceedings of the Institution of Civil Engineers – Maritime Engineering* 161, 9–21.

- Tyrrell, M.C., Byers, J.E., 2007. Do artificial substrates favor nonindigenous fouling species over native species? *Journal of Experimental Marine Biology and Ecology* 342, 54–60.
- UK Biodiversity Group, 1999. UK Biodiversity Group Tranche 2 Action Plans – Volume V: Maritime Species and Habitats. English Nature, Peterborough, http://www.ukbap.org.uk/Library/Tranche2_Vol5.pdf (accessed March 2011).
- van der Wal, D., Pye, K., 2004. Patterns, rates and possible causes of saltmarsh erosion in the Greater Thames area (UK). *Geomorphology* 61, 373–391.
- Vaselli, S., Bulleri, F., Benedetti-Cecchi, L., 2008. Hard coastal-defence structures as habitats for native and exotic rocky-bottom species. *Marine Environmental Research* 66, 25.
- Walker, S.J., Schlacher, T.A., Thompson, L.M.C., 2008. Habitat modification in a dynamic environment: the influence of a small artificial groyne on macrofaunal assemblages of a sandy beach. *Estuarine, Coastal and Shelf Science* 79, 24–34.
- Walsh, K.J.E., Betts, H., Church, J., Pittock, A.B., McInnes, K.L., Jackett, D.R., McDougall, T.J., 2004. Using sea level rise projections for urban planning in Australia. *Journal of Coastal Research* 20, 586–598.
- Webster, P.J., Holland, G.J., Curry, J.A., Chang, H.R., 2005. Changes in tropical cyclone number, duration, and intensity in a warming environment. *Science* 309, 1844–1846.
- Weigel, R.L., 2002a. Seawalls, seacliffs, beachrock: what beach effects? Part 1. *Shore and Beach* 70 (1), 17–27.
- Weigel, R.L., 2002b. Seawalls, seacliffs, beachrock: what beach effects? Part 2. *Shore and Beach* 70 (2), 13–22.
- Weigel, R.L., 2002c. Seawalls, seacliffs, beachrock: what beach effects? Part 3. *Shore and Beach* 70 (3), 2–14.
- Wood, D.W., Bjørndal, K.A., 2000. Relation of temperature, moisture, salinity and slope to nest site selection in loggerhead sea turtles. *Copeia* 2000, 119–128.
- Yapp, G.A., 1986. Aspects of population, recreation, and management of the Australian coastal zone. *Coastal Zone Management Journal* 14, 47–66.
- Zaikowski, L., McDonnell, K.T., Rockwell, R.F., Rispoli, F., 2008. Temporal and spatial variations in water quality on New York South Shore Estuary tributaries: Carmans, Patchogue, and Swan Rivers. *Estuaries and Coasts* 31, 85–100.