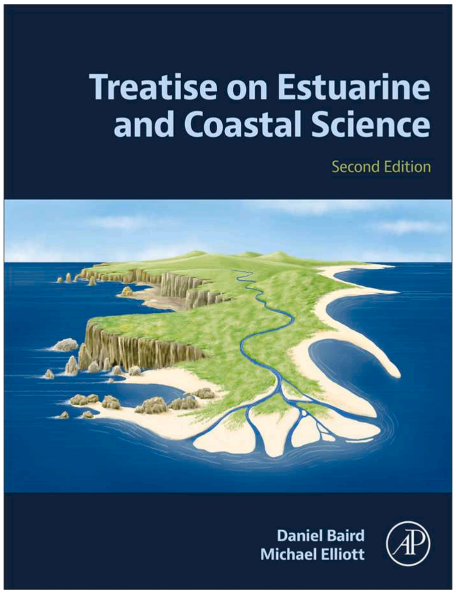

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6.2 Estuarine and Coastal Structures: Environmental Effects and a Focus on Shore and Nearshore Structures

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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, although, more recently, there has been increasing emphasis on biologically softening armored shorelines using green infrastructure, nature-based and living shoreline approaches. Despite widespread use on all types of shorelines, information regarding the scale of ecological impacts of shoreline armoring continues to be quite limited relative to the number and extent of new structures proposed globally. Here, we summarize existing knowledge on the effects of armoring structures on the biodiversity, productivity, structure, and function of coastal ecosystems.

Key Points

- This chapter reviews the ecological effects of shoreline armoring on coastal marine ecosystems.
- Shoreline armoring is a prevalent and widespread response to the risks to human infrastructure from erosion, storms, flooding, climate change and sea-level rise.
- Armoring structures are placed in a wide range of coastal soft sediment ecosystems, including marshes, estuaries, river mouths, harbors and sandy beaches.
- Armoring structures vary greatly in type, purpose, size, construction, and age; all of these characteristics affect ecological impacts to coastal ecosystems.
- The ecological impacts and responses to coastal armoring are discussed, including effects on the quantity and quality of intertidal and shallow subtidal habitats, biotic communities, populations, food webs, higher trophic levels, biodiversity, invasive species and ecosystem functioning.
- Significant ecological impacts of armoring ranging from habitat loss to declines in biodiversity and function have been reported across all soft sediment ecosystem types.
- Progress toward generalizing the ecological effects of shoreline armoring structures on coastal ecosystems, includes the projection of greater impacts for:
  a. upper intertidal zones compared to mid and lower intertidal zones
  b. armoring structures designed to stop water flow (e.g., seawalls and bulkheads) compared with those intended to slow water flow (e.g., sills).
- Climate change will exacerbate the impacts of existing shoreline armoring structures, escalate hazards and structural failures and increase construction of new armoring structures in many regions.
- Substantial ecological and ecosystem benefits can be realized when shoreline armoring structures are removed or shifted landward to reduce interactions with waves and tides and increase habitat space.
- Developing ecosystem-based adaptation approaches that combine risk and hazard mitigation and the conservation of soft sediment ecosystems in coastal zones could provide co-benefits and a path forward for society.

6.2.1 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 (Blankespoor *et al.*, 2014; Nordstrom, 2000; Slott *et al.*, 2006; Vousdoukas *et al.*, 2020; Reed *et al.*, 2022).

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. Most 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 pose severe threats to beaches, coastal wetlands, and river deltas (Adam, 2002; Morris *et al.*, 2002; Schlacher *et al.*, 2007; Vitousek *et al.*, 2017). On many 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 man-made barriers, coastal ecosystems are increasingly caught in a coastal squeeze between rising seas and expanding development (Doody, 2004; Schlacher *et al.*, 2007; Pontee, 2013) (**Fig. 1**). Projections of habitat loss relative to possible future changes in sea level, recession of coastlines, and coastal squeeze are available for a growing number of coastal areas and habitats (e.g., Borchert *et al.*, 2018; Kirwan and Megonigal, 2013; Myers *et al.*, 2019; Vitousek *et al.*, 2017). 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; Vitousek *et al.*, 2017), and for some regions tipping points for intertidal zones have already been reached (Törnqvist *et al.*, 2020; Barnard *et al.*, 2021). 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 before the end of the century (Nicholls *et al.*, 1999), although uncertainty is considerable. For example, a loss of about 4000 ha of freshwater and brackish habitats was predicted for the UK as a consequence of the combined effects of sea-level rise and coastal development (Lee, 2001).

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

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, coastal armoring structures are not a recent phenomenon, although their extent and size have increased dramatically in recent decades, a trend that is projected to intensify. For example, there has been extensive armoring of more than 60% of Singapore Harbour (Perkins *et al.*, 2015) and of more than 300 km of shoreline in Chile (Aguilera, 2018) in the past 20 years. Despite growing evidence and awareness of the impacts of the substantial environmental impacts of coastal cities and infrastructure, major changes to coastlines continue unabated (Perkins *et al.*, 2015; Evans *et al.*, 2017, 2019). However, detailed and applicable information on either the extent of changes or the specific ecological effects of such changes are not available for many regions (Bulleri, 2006; Airoldi and Beck, 2007; Chapman *et al.*, 2018; Strain *et al.*, 2020; Dodds *et al.*, 2022; Gittman *et al.*, 2015, 2016b; Dugan *et al.*, 2018; Floerl *et al.*, 2021).

6.2.2 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,b; Baxter *et al.*, 2023). Early forms of armoring included breakwaters and other structures built to stabilize harbors along Mediterranean coasts by 2 BCE, and large coastal defense projects initiated in China by 25 BCE (Charlier *et al.*, 2005). As early as 175 BCE, 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; Airoldi *et al.*, 2005a; Gittman *et al.*, 2015; Floerl *et al.*, 2021). 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; Airoldi *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, it 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; Lumbroso and Vinet, 2011) with risks extending beyond infrastructure to human safety. The ongoing need to monitor, repair, and maintain coastal armoring structures, is costly. Much of the responsibility for maintaining or building seawalls, groynes, and other coastal defense structures belongs to government authorities, for which repairs and replacement of seawalls can form a major part of their budget (Nordstrom, 2014); however, in some countries many kilometers of aging seawalls are orphaned with little or no oversight (Lumbroso and Vinet, 2011). These costs are ultimately, one way or another, passed on to the general public.

6.2.3 Types of Structures

Design and engineering of coastal armoring structures, and the materials used to build them, vary widely, as do their construction costs, efficacy, and life span (Nordstrom, 2014; Perkins *et al.*, 2015; Dodds *et al.*, 2022). 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.

Fig. 2 An intertidal concrete seawall located on a beach along the open coast Santa Barbara, California, USA.

6.2.3.1 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 (**Fig. 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 (**Fig. 3**). However, along sheltered shores, such as estuaries and tidal channels, bulkheads can also be placed lower on the shore to act as a primary coastal defense (**Fig. 4**) (Nordstrom, 2000).

Revetments, by contrast, are mostly constructed 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 and width (**Fig. 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' (**Fig. 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; **Fig. 6**). Jetties or breakwaters that extend out from the shoreline at inlets or harbor mouths are used to control the flow of water and 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 (**Fig. 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 may contain several different types of artificial structures that can be very extensive, covering hectares and providing berths for hundreds of yachts and small vessels (**Fig. 7**).

Developers have responded to the increased demands for waterfront properties by creating 'canal estates' in sheltered waters (**Fig. 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).

Fig. 3 A bulkhead built to protect coastal development on a sheltered shoreline, Savannah, Georgia, USA.

Fig. 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.

6.2.3.2 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

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

Fig. 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.

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 (Fig. 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

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

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.

6.2.3.3 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 defense 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; Heery *et al.*, 2017). 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 is still relatively 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 consideration of the resulting ecological impacts on coastal ecosystems, or how these structures may affect biodiversity, productivity, and the provision of ecosystem functions (Dugan *et al.*, 2018; Heery *et al.*, 2017; Evans *et al.*, 2019). Here, we summarize accessible information on the known extent of these structures for regions where sufficient data are available in Europe, North America, Australia, and Asia. Using available information, Bugnot *et al.* (2021) estimated that the footprint of built structures covered at least 32,000 km^2 worldwide as of 2018, and that value is expected to increase. However, an important caveat they noted is that the extent of armoring structures is poorly quantified for many coastal regions. This, and the fact that armoring is increasing in many regions, means that the summaries cited here represent only a snapshot in time and are highly likely to be an underestimate of the extent of coastal armoring in the world. However, they provide a starting point for evaluating the scope of putative environmental effects of shoreline armoring on coastal ecosystems.

In Europe, $>$ 15,000 km of the coastline is actively retreating, despite coastal protection works along 2900 km, while another 4700 km of coast is artificially stabilized (EC, 2004). A review of the status of European coastlines (Airoldi and Beck, 2007 and references therein) has shown that 22,000 km^2 of the European coastal zone was 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 Airoldi 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 (**Fig. 9**), make up more than 60% of the coastline along 300 km of naturally low sedimentary shoreline (Bacchiocchi and Airoldi, 2003). Similarly, in Belgium and in the Wadden Sea region, there are no natural

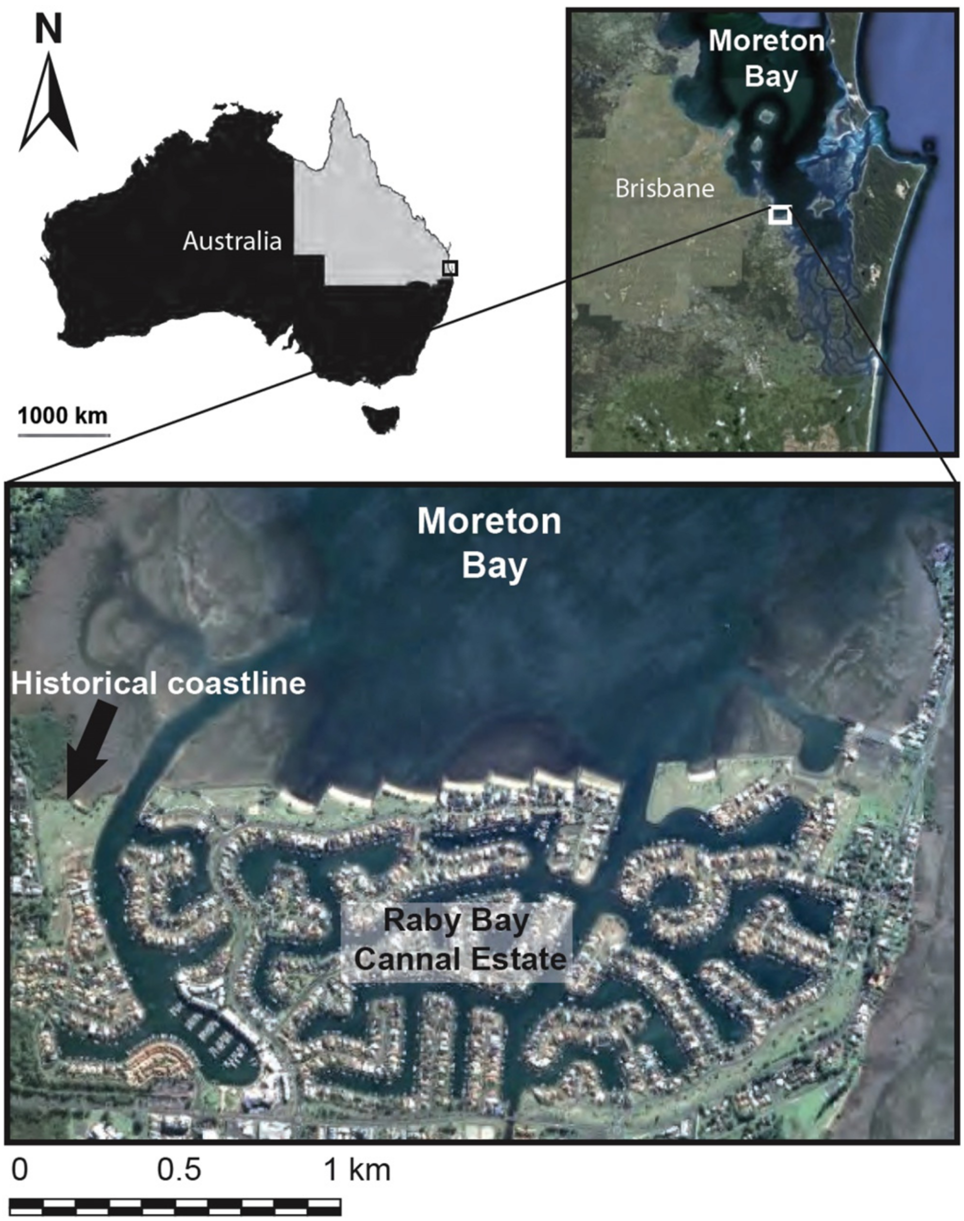

Fig. 8 An example of the development of canal estates near Brisbane, Australia.

rocky shores, and all intertidal hard substrata are created by man-made structures (Johannesson and Warmoes, 1990; Reise, 2005). Coastal urbanization is projected to 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. A review by Gittman *et al.* (2015) estimated that 22,842 km or ~14% of the shoreline of the continental USA was armored overall. However, the proportion of armored shoreline varies greatly with region, shoreline exposure and wave height (Gittman *et al.*, 2015). 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 (Erdle *et al.*, 2008). 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, 1990). Trends are similar for the Pacific coastline in California, where the geographic extent of armoring on the 1763 km coast increased by 400% between 1971 (2.5%) and 1992 (12%) (Griggs, 1998) and had reached 13.9% by 2018 (Griggs and Patsch, 2019). 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 Foundation, 2010). However, farther north, on the sheltered shores of the Puget

Fig. 9 Aerial view of the urban structures along the coasts of the Adriatic sea in northern Italy. Photo by Benelli, reproduced from Airoldi, 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.

Sound in Washington State, armoring covers > 30% of the coastline (Munsch *et al.*, 2015). These patterns are also evident in the Western Pacific. In Japan, 15,900 km of the 34,500-km coastline were 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). For Okinawa Island, 63% of the coastline has been altered (Masucci and Reimer, 2019). Most of the population of Australia is concentrated in 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

Fig. 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.

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 (**Fig. 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).

6.2.4 Current State of Knowledge on Environmental Effects

6.2.4.1 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; Nordstrom, 2014). For soft-sediment habitats, the loss of original habitat that is covered by the footprint of man-made coastal structures can be 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; Heery *et al.*, 2017). 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,b). 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,b,c; Griggs, 2005b; Heery *et al.*, 2017). 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 (**Fig. 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 (**Fig. 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

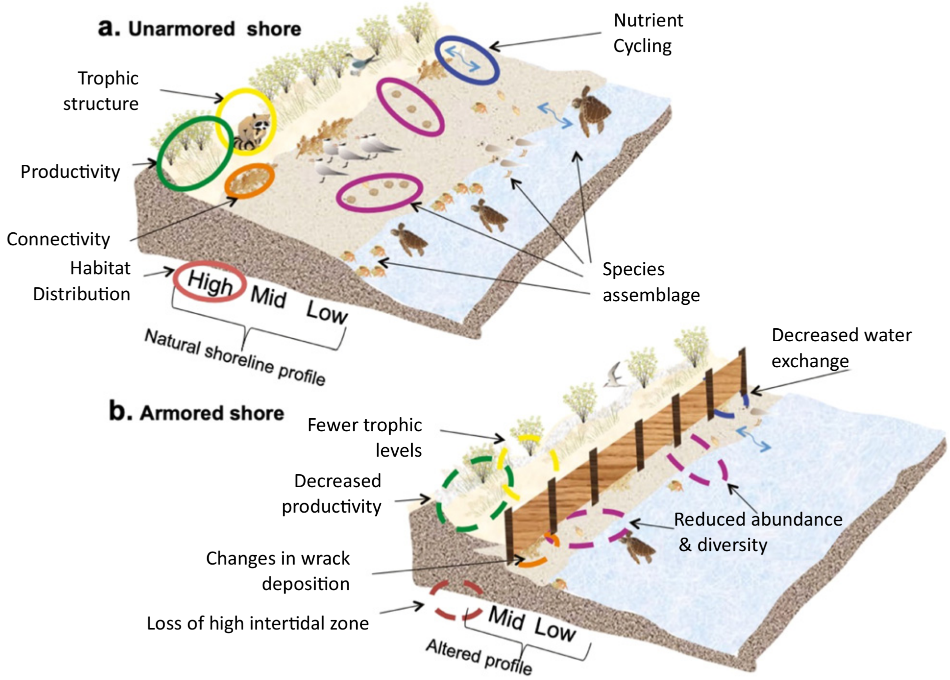

Fig. 11 Illustration comparing a hypothetical unarmored (a) with an armored (b) shoreline, with examples for six of the potential ecological responses to armoring: habitat distribution, species assemblage, trophic structure, nutrient cycling, productivity, and connectivity. Broken ellipses in panel b signify negative impacts and correspond to the ellipse of the same color in panel a. Adapted from Dugan *et al.* (2018).

increased wave reflection and the narrowing of the surf zone during storms (e.g., Hall and Pilkey, 1991; Griggs, 1998, 2005a,b; 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,b,c).

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 (O'Meara *et al.*, 2015). 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; O'Meara *et al.*, 2015).

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). To offset impacts on sediment transport and downcoast erosion, the height of groynes has been reduced 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). Inlet or harbor mouths that have been stabilized with jetties often require regular or, in regions with strong littoral currents, continuous dredging or bypassing to move trapped sediments across the entrances (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 these structures on open coasts (**Fig. 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 also block the 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

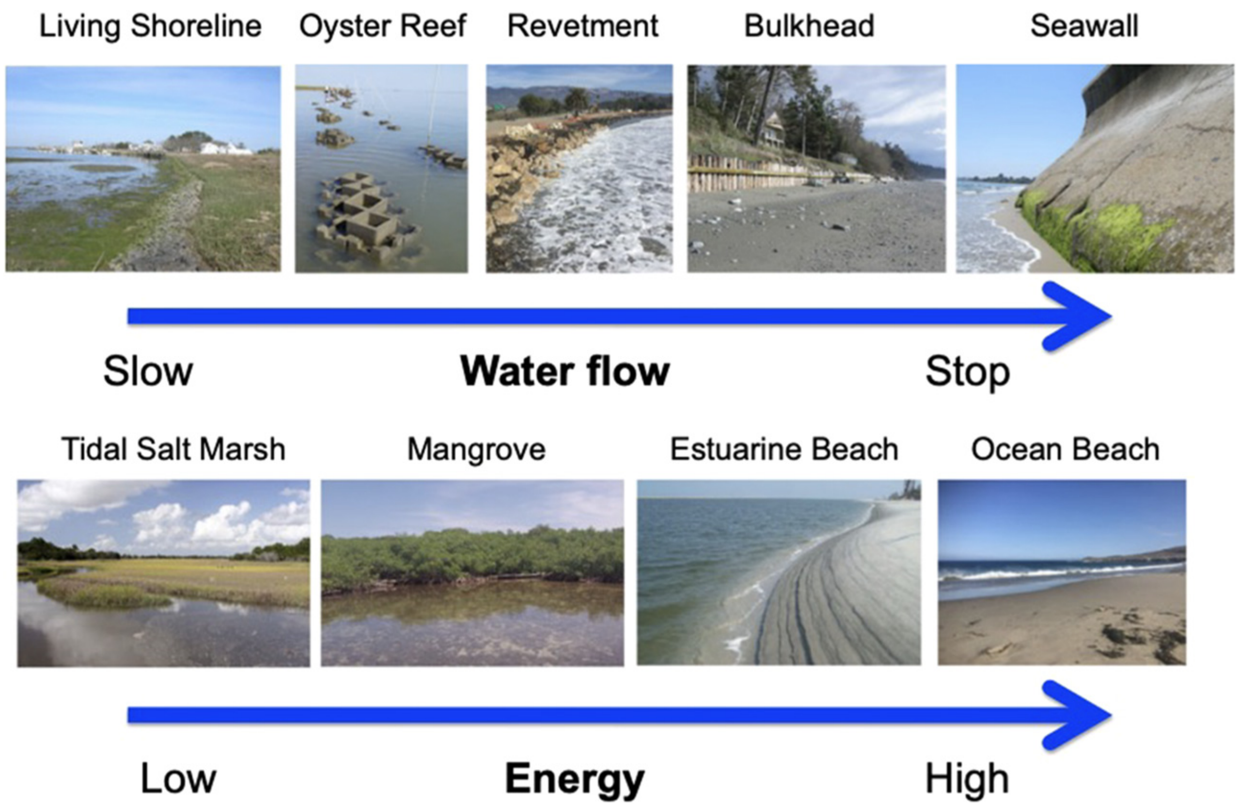

Fig. 12 Illustration of gradients in the two axes of influence for the conceptual model of shoreline armoring effects. Top row: Engineering purpose with respect to intended effect of structure on water flow (slow vs. stop). Bottom row: Hydrodynamic energy (low to high) at the structure as determined by the environment. Adapted from Dugan *et al.* (2018).

accumulation of fine sediments and organic matter, creating stagnant conditions typical of lagoons (Martin *et al.*, 2005). However, detached offshore structures are not common in the more sheltered waters of estuaries.

6.2.4.2 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, information on the ecological effects of these structures had lagged (NRC, 2007) until recent years. However, ecological effects of armoring are gradually becoming more clearly understood and described in the literature. In soft sediment environments, ecological responses to armoring can manifest in effects on habitat distributions, species assemblages including wildlife and fisheries, productivity, trophic structure, connectivity and functioning such as nutrient cycling (**Fig. 11**) (Dugan *et al.*, 2018). Ecological impacts of armoring can affect more than soft sediment biota (e.g., Heery *et al.*, 2017; Jaramillo *et al.*,2012, 2021), with effects reported for important fish populations (Munsch *et al.*, 2015), sessile invertebrates occupying rocky substrata (Chapman, 2003; Gittman *et al.*, 2016b) and the spread of non-native species (Dong *et al.*, 2016; Dafforn, 2017).

As a consequence of this increasing knowledge, ecological impacts are more often being considered in policy decisions regarding coastal protection. However, many of the ecological impacts that have been documented for armoring are site-specific. Consequently, there is a need to synthesize results of studies over larger spatial or temporal scales (but see Dugan *et al.*, 2018; Gittman *et al.*, 2016b; Heery *et al.*, 2017; Evans *et al.*, 2019). At the same time, some authors have cautioned against generalizing from limited field data (Bulleri and Chapman, 2010; Firth *et al.*, 2020) and spatial scales (Critchley and Bishop, 2019; Critchley *et al.*, 2021). 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 and scales, on coastal ecosystems is increasingly urgent.

To synthesize the current understanding of the ecological impacts of armoring and coastal defense structures and to identify urgent research needs, we review emerging conceptual information, broader analyses, and a subset of case studies. 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.

The wide variety of types, sizes, and placements of armoring structures that have been built across an array of coastal ecosystems and contexts have made general predictions concerning the ecological effects of armoring challenging. To address this issue, a conceptual model of armoring impacts was developed to scale predicted ecological effects of shore-parallel armoring across

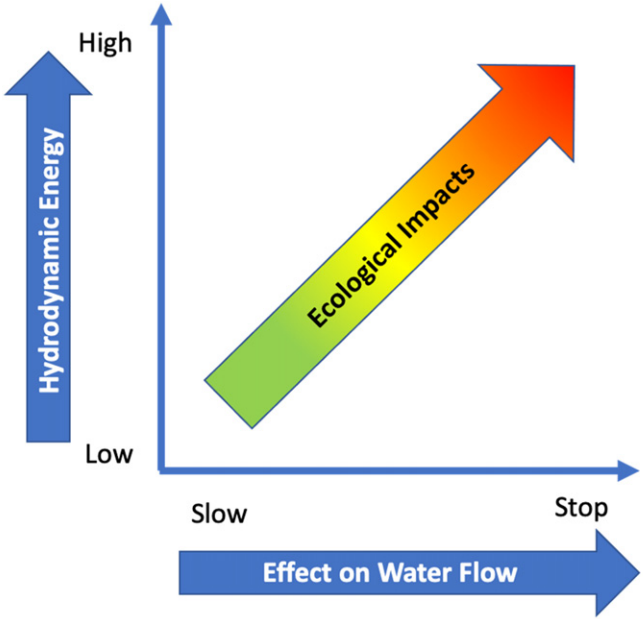

Fig. 13 Conceptual model showing predicted ecological impacts of shoreline armoring in soft sediment environments across the array of structure types used to either slow or stop water flow (x-axis) and with different hydrodynamic energy levels at the armoring structure (y-axis). Ecological impacts are predicted to increase as one moves up and to the right within the parameter space. Adapted from Dugan *et al.* (2018).

a wide range of soft sediment habitats and armoring structures (Dugan *et al.*, 2018). The model used two primary axes: (1) engineering purpose with regard to water flow (reduce/slow flow or prevent/stop flow) of the structure; and (2) hydrodynamic energy of the environment in terms of tides, currents and waves (**Fig. 12**) (Dugan *et al.*, 2018). Along the purpose/water flow axis, armoring structures designed to slow rather than stop water such as living shorelines and sills were expected to allow more natural functioning and connectivity than those designed to stop water flow to the shoreline, such as seawalls and bulkheads. Along the hydrodynamic energy axis, armoring structures built in higher energy environments, such as wave exposed open coasts were expected to have greater impacts than those in low energy settings . In summary, the ecological effects of armoring structures were expected to increase with structures designed to stop as opposed to slow water flow and with increasing hydrodynamic energy of the environment of the structure (**Fig. 13**). Thus, an armoring structure designed to slow water flow in a low energy environment (e.g., a low crested sill in a tidal marsh) was expected to have the lowest ecological impacts while one designed to stop water flow in a high energy environment (e.g., seawall on an open coast beach) would have the greatest ecological impacts. Evaluation of this conceptual model using results of a literature review indicated a variety of ecological responses to armoring varied most clearly with the engineering purpose of structures, with a higher frequency of negative responses for structures designed to stop water flow within a given hydrodynamic energy level. Comparisons across the hydrodynamic energy axis were less clear-cut, but negative ecological responses prevailed (>78%) in high-energy environments (Dugan *et al.*, 2018).

6.2.4.2.1 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 a rocky reef.

The effects of armoring on habitat are relatively well studied, but the majority of studies have been conducted in low energy environments (see Dugan *et al.*, 2018). Overall, Dugan *et al.* (2018) found the majority of studies reported negative impacts of armoring on habitat, generally citing loss of habitat and reduced habitat quality, particularly for structures designed to stop water flow. However, for nature-based designs, such as oyster reefs and living shorelines, impacts on habitat were relatively less, and more positive results were reported. For sheltered coasts of 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 intertidal and estuarine species (Peterson *et al.*, 2000; Bilkovic *et al.*, 2006; Pérez-Ruzafa *et al.*, 2006;

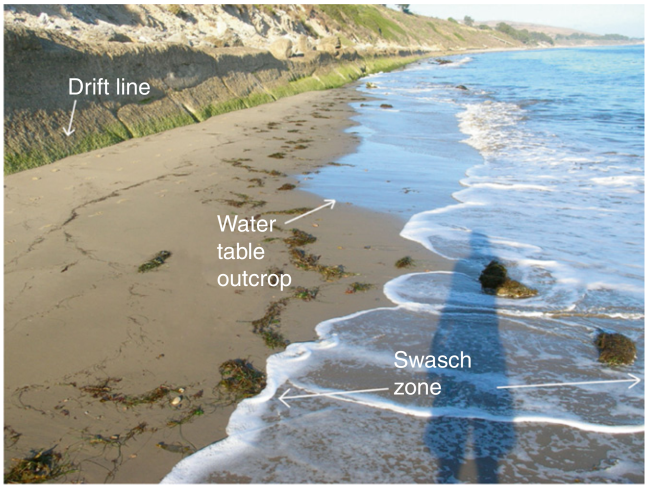

Fig. 14 This view looking east along an old concrete seawall on the Gaviota coast of California, USA, at low tide illustrates the attenuation of ecologically important intertidal zones on a beach seaward of coastal armoring. Adapted from Dugan, J.E., Hubbard, D.M., 2006. Ecological responses to coastal armoring on exposed sandy beaches. Shore and Beach 74 (1), 10–16.

Seitz *et al.*, 2006; NRC, 2007; Toft *et al.*, 2007; Bilkovic and Roggero, 2008; Bilkovic and Mitchell, 2013). 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 wide variety of forms and applications of coastal armoring, these studies did not account for the relative responses of the different habitat zones of the beach (e.g., McLachlan and Jaramillo, 1995), 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 projected using changes in the distributions of the different habitat zones of the beach as proxies for habitat loss (**Fig. 14**, **Table 1**) (Dugan and Hubbard, 2006; Dugan *et al.*, 2008; Jaramillo *et al.*, 2012, 2021). As the width of the overall beach and intertidal zone becomes narrower 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 on habitat are predicted to be 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 where present (Dugan *et al.*, 2008; Dethier *et al.*, 2016). Habitat near the drift line, the primary zone for wrack-associated invertebrates, may also be greatly reduced or eliminated (Dugan and Hubbard, 2006; Dugan *et al.*, 2008; Heerhartz *et al.*, 2014; Dethier *et al.*, 2016; Jaramillo *et al.*, 2012, 2021). As the drift line habitat shifts from the sandy 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 comparative studies of armored and unarmored beaches to date support this framework, which can also be applied to the more sheltered shores of bays and estuaries with some modification (Dethier *et al.*, 2016; Heerhartz *et al.*, 2014, 2016). The scale of habitat and ecological effects of armoring was observed to be strongest for the upper shore zones in studies of seawalls along undeveloped open coastlines in Chile and California (Dugan and Hubbard, 2006; Dugan *et al.*, 2008; Jaramillo *et al.*, 2021) and in the sheltered waters of Puget Sound (Sobocinski *et al.*, 2010). In a study of more than 60-year old seawalls on open-coast beaches, there were no high beach zones (above the drift line) remaining on the armored segments compared to adjacent unarmored segments where those zones, albeit narrow, averaged 3.5 m in width (Dugan and Hubbard, 2006; Dugan *et al.*, 2008). This finding was consistent with the scale of placement loss expected for the seawall type studied and demonstrated the relative ecological importance of this impact for narrow beaches. The overall narrowing of the armored beaches observed above the water table outcrop on armored segments (average 11.4 m) was, however, much greater than that expected from placement loss, suggesting the effects of passive erosion on the study beaches (Dugan *et al.*, 2008).

Another important consideration relative to the generality of these predictions is the location of the armoring structure on the beach profile, which affects the hydrodynamic setting (see Dugan *et al.*, 2018, Jaramillo *et al.*, 2021), namely the amount of interaction with waves and tides and the resulting physical impacts (Weigel, 2002a,b,c). Habitat loss is expected to scale with the intensity of interaction between structures and coastal processes (e.g., wave reflection and tidal action) (Dugan *et al.*, 2018) (**Fig. 13**), which is predicted to increase as the structure ages, coasts erode, 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, subsequent erosion of the shore or increases in sea level.

Loss of hard natural habitat, such as rocky reefs, has generally been considered to be less of a problem (Thompson *et al.*, 2002), because artificial structures may act as suitable surrogate habitat. Although this may be true for some subtidal taxa, which settle in

Table 1 Hypothesized ecological responses to alongshore armoring on open coast 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.

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 the loss of 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 and zones 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).

6.2.4.2.2 Alteration of ecological structure, function, and integrity

6.2.4.2.2.1 Coastal vegetation

Intact coastal vegetation, including mangroves, salt marshes, seagrasses, and macroalgae, as well as coastal strand and dunes, buffer shores and retain sediments from the effects of erosive processes, such as tides, waves, and storms. These communities and structures provide valuable ecosystem functions including primary production, water filtration, uptake of nutrients, detrital production, and degradation and carbon fixation (Costanza *et al.*, 1997). 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).

Shoreline vegetation and primary production are often lost from open and sheltered habitats where bulkheads and seawalls both directly cover and alter habitat and prevent the migration of the shoreline in response to the changing sea level (**Fig. 11**). In the United Kingdom, for example, 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). In mangrove ecosystems, seawalls were associated with reduced area of mangrove forest habitats (Heatherington and Bishop, 2012). Submerged aquatic vegetation (SAV) was negatively affected by bulkheads in Chesapeake and other west Atlantic bays (Patrick *et al.*, 2016).

In salt 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 greater effects, depending on the amount of tidal flushing.

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; Rodil *et al.*, 2015) due to a lack of sufficient upper beach habitat that restricts the resulting ecotone between intertidal beach and the coastal strand and dune zone. 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 and foredunes on coastlines that are either retreating or developed or both.

6.2.4.2.2.2 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; Bilkovic and Mitchell, 2013) (**Fig. 11**). 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). In a mangrove ecosystem, the addition of a seawall reduced availability and prevented movement of propagules (Anthony and Gratiot, 2012).

6.2.4.2.2.3 Wrack and connectivity

Along with habitat loss, the alteration of physical processes that affect the deposition and retention of sediments and erosion signals 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 subsidies for biota as food or habitat or both (e.g., Colombini and Chelazzi, 2003; Dugan *et al.*, 2003; Hyndes *et al.*, 2022 **Table 1**: **Fig. 11**). The effect of armoring on wrack retention on beaches could significantly impact the functioning of open-coast beach ecosystems through effects on coastal nutrient cycling and dynamics (Dugan *et al.*, 2011; O'Meara *et al.*, 2015). 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 (Dugan and Hubbard, 2006) and protected beaches of Puget Sound (Sobocinski *et al.*, 2010; Dethier *et al.*, 2016; Heerhartz *et al.*, 2016) that reported significantly lower standing stocks of wrack and driftwood on armored beaches compared to natural beaches. However, in one study of a variety of mangrove ecosystems, wrack abundance was not significantly affected by armoring (Critchley *et al.*, 2021).

In 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 and breakwaters can also potentially trap higher accumulations of anthropogenic litter, macrophyte wrack, and terrestrial detritus delivered by littoral currents causing a variety of potential impacts (Aguilera *et al.*, 2016; Strain *et al.*, 2018). 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).

6.2.4.2.2.4 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. Effects on species assemblages were the most commonly documented ecological response to shoreline armoring with the majority of studies conducted in low energy environments (see review by Dugan *et al.*, 2018). Armoring was associated with declines in species diversity and abundance across all soft sediment environments and structure types (Dugan *et al.*, 2018). However, negative responses of species assemblages to armoring prevailed for structures designed to stop water flow while some positive responses were reported for structures designed to slow water flow (Dugan *et al.*, 2018) (**Fig. 13**).

For sheltered shores and estuaries, the predicted ecological impacts on intertidal and subtidal benthic fauna from the loss of intertidal and shallow-water zones and depth gradients associated with shoreline armoring and bulkheads are broadly supported by the results of recent individual studies (Morley *et al.*, 2012; Lawless and Seitz, 2014; Dethier *et al.*, 2016; Heerhartz *et al.*, 2016), meta-analyses (Gittman *et al.*, 2016b) and reviews (Dugan *et al.*, 2018). The density and diversity of subtidal benthic bivalves, which comprise more than 50% of benthic 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

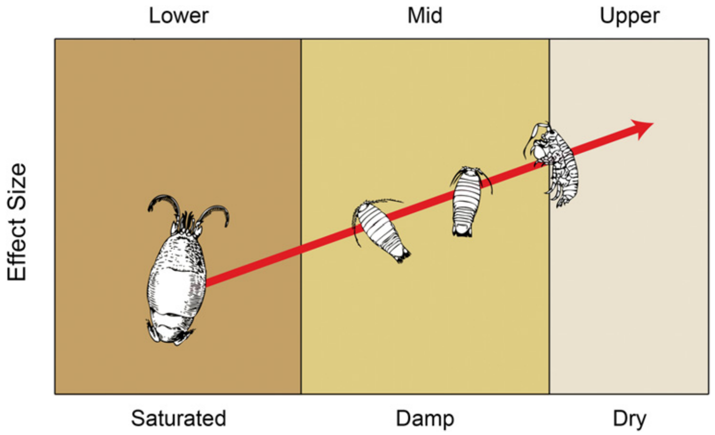

Fig. 15 Hypothesized scaling of effect sizes for coastal armoring impacts on important mobile macroinvertebrates (Anomuran decapods, cirolanid isopods and talitrid amphipods) from the lower to the upper intertidal zones of sandy beaches in the eastern Pacific. Effect size of impacts is projected to increase from the lower to the upper intertidal zone. Adapted from Jaramillo *et al.* (2021).

shores (Seitz *et al.*, 2006). However, these impacts have been reported to differ for some sheltered and muddy shores (Critchley and Bishop, 2019).

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). 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). 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.

As indicated above, the impacts of shore-parallel armoring are less well studied on open coasts, than on sheltered shores and very few community-level analyses have been conducted for sandy beaches (see review by Heery *et al.*, 2017 for further details). Although more research is needed, the prediction of the effect sizes for impacts of armoring increasing from lower to upper shore biota (**Fig. 15**) is generally supported by results for intertidal invertebrates of sandy beaches (Dugan *et al.*, 2008; Jaramillo *et al.*, 2021 and studies detailed below). For open-coast beaches in California and Chile, the abundance and biomass of mobile upper shore invertebrates were significantly greater on unarmored beach segments than on armored segments with the largest effect sizes relative to mobile invertebrates of the mid and lower shore (Jaramillo *et al.*, 2021). For protected shores in Puget Sound, the abundance of talitrid amphipods and insects was also significantly higher on natural beaches than on armored beaches (Sobocinski *et al.*, 2010). Toft *et al.* (2007) reported a similar decline in high-shore benthos in Puget Sound and also suggested that the reduction in food resources related to the loss of shoreline vegetation may have contributed. Likewise, the abundance of upper intertidal beetles was significantly lower on armored beaches in Brazil (Laurino *et al.*, 2022). On beaches in Australia, the densities of burrows of the upper shore 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). Collectively these results are consistent with the expected strength of ecological responses of intertidal biota to a loss of upper shore habitat on armored beaches.

On open-coast beaches, the distribution and survival of mobile invertebrates of the mid and lower shore (e.g., cirolanid isopods, 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.*, 2021) imposed by armoring structures. The regular movement of these mobile intertidal animals to adjust to waves and tides (Jaramillo *et al.*, 2002a; Dugan *et al.*, 2013) may intensify the ecological impacts of armoring. For example, the restriction of tidally generated landward migration of a mid-intertidal cirolanid isopod, *Excirolana chiltoni*, imposed by a seawall, was illustrated by Klapow (1972) (**Fig. 16**). 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. Studies of responses of mid- and lower beach invertebrates to shoreline armoring are few. However the expected scaling of armoring impacts for these biota was evident in results of surveys in Chile and California designed to examine this question (**Fig. 15**) (Jaramillo *et al.*, 2021). In that study, the abundance of mid-intertidal zone invertebrates (two isopod species) was significantly lower in the presence of seawalls and rock revetments, and these taxa could be eliminated from some armored beaches. However, effect sizes were lower than observed for upper beach invertebrates. As expected, the lower intertidal macroinvertebrates (hippid crabs) were generally the least affected by coastal armoring; however, where armoring structures were located low on the beach profile, even low intertidal invertebrates were negatively affected (Jaramillo *et al.*, 2021). These results illustrate

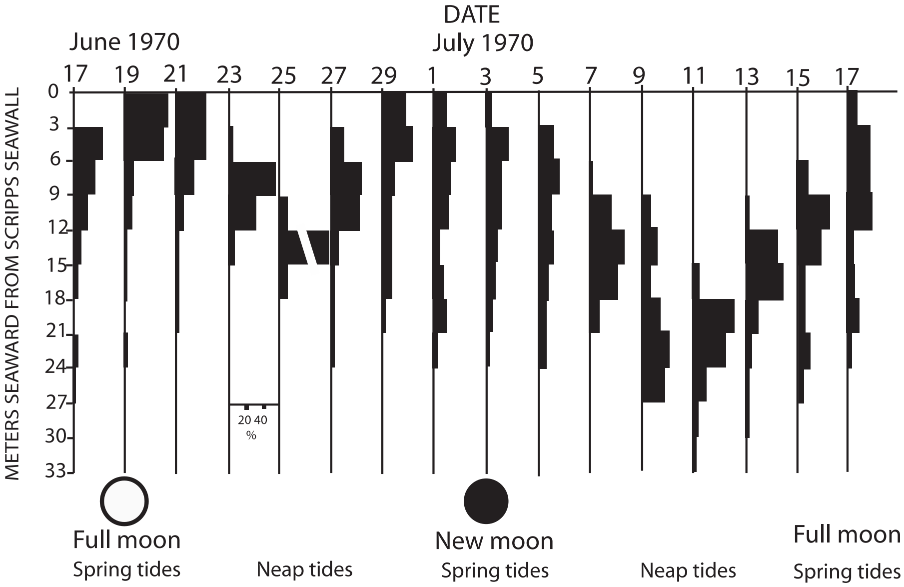

Fig. 16 Semimonthly changes in the position of the intertidal zone occupied by the mobile beach isopod, *Excirolana chiltoni*, seaward of a seawall on the beach at Scripps Institution of Oceanography, San Diego, California during summer 1970. On spring tides, especially in June and early July, the wave 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.

the importance of factors that affect the hydrodynamic energy at the structure (**Fig. 12**), such as the age of the structure and its resulting elevation on the beach profile, in projecting both habitat loss and ecological impacts. Results of a short term (<2 year) study of a newly constructed seawall on an open-coast beach that found no significant effects on intertidal invertebrates (Jaramillo *et al.*, 2002b) also suggest that ecological impacts to biota and habitat may accrue with the age of the structure and the evolution of the beach and environment, with impacts increasing over time.

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.

In many parts of the globe, armoring of coastal areas with groynes is far more extensive, with numerous structures placed in close proximity (**Fig. 6**) (Bush *et al.*, 2001; Fanini *et al.*, 2009) and potentially greater ecological impacts. Walker *et al.* (2008) showed that 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).

6.2.4.2.2.5 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, like many other studies on armored shorelines, results often vary greatly from place to place. 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 (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 shading 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 (2008a) 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, 2008b). 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. Recent studies have shown that different artificial structures influence the behavior, feeding and abundances of different fish species and life stages in complex ways (Munsch *et al.*, 2014, 2015; Heerhartz and Toft, 2015). However, more research is needed to understand the widespread and long-term effects of armored shorelines on fish populations.

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 assessed.

Some ecological effects of coastal armoring may involve interactions with human activities, such as fishing. 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 of fish from surrounding areas or increased production leading to larger populations.

A further consideration is that artificial structures may provide important habitat and protection for some threatened species that may be experiencing loss of natural habitat, such as seahorses (Clynick, 2008b; Simpson *et al.*, 2019). This sort of interaction creates additional complexity in the management of shorelines altered by artificial structures.

6.2.4.2.2.6 Barriers to movement of animals and wrack

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

6.2.4.2.2.7 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 a variety of land birds (Hubbard and Dugan, 2003; DeLuca *et al.*,

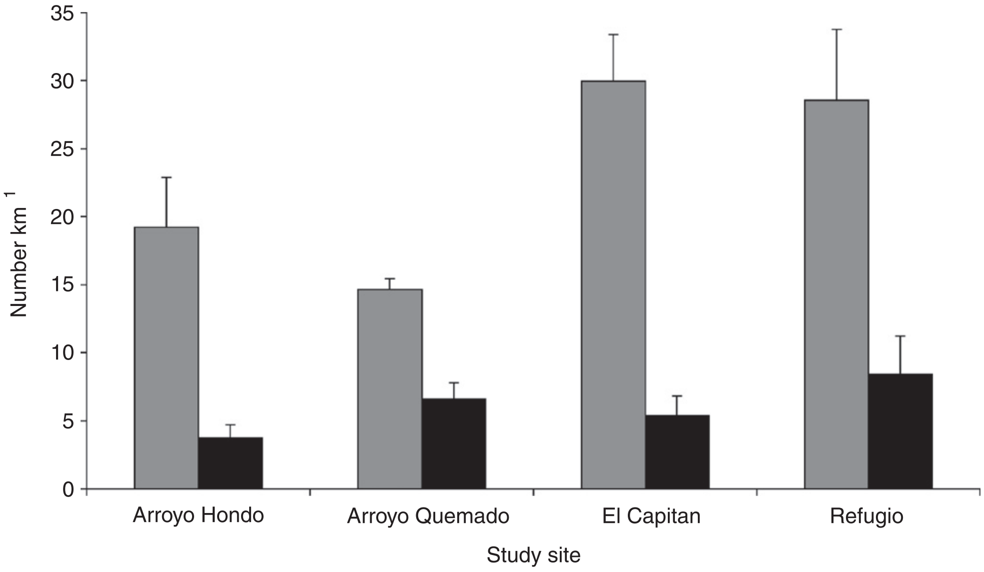

Fig. 17 Comparison of the abundance of shorebirds (Mean values + one standard error, $n = 8$) during fall migration for four pairs of adjacent bluff-backed (gray bars) and armored (black bars) segments of sandy beach in August and September 2005 in California (From Dugan and Hubbard, 2006). Average abundance of these birds was 28% lower on beach segments next to armoring structures.

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). Shorebird diversity and abundance have been correlated with prey availability for sheltered soft sediment habitats (Colwell and Landrum, 1993; Placyk and Harrington, 2004; VanDusen *et al.*, 2012) and 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 (**Fig. 11**). For example, at subestuary scales in the Chesapeake Bay, shoreline hardening with bulkheads and riprap negatively affected habitat use by post-breeding, migratory and wintering birds, including shorebirds and waterfowl (Prosser *et al.*, 2018). In heavily armored Delaware Bay, declines in horseshoe crab eggs in have been associated with reduced body weights and abundance of migratory red knots (Niles *et al.*, 2009). 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) suggest that ecological impacts on coastal avifauna can be substantial (**Figs. 17**, **18**). Of note, the significant effects of armoring on birds have been 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), suggesting that other factors, including prey abundance, availability of high-tide feeding habitat and refuges, 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 have exceeded that of the loss of habitat, 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 (**Fig. 17**), but also by roosting birds, such as gulls, pelicans, and other seabirds (**Fig. 18**), indicates that impacts of armoring beyond foraging opportunities 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), Atlantic silverside *Menidia menidia* (Balouskus and Targett, 2012), and sand lance *Ammodytes hexapterus* (Reeves *et al.*, 2003; Huard *et al.*, 2022), 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 reducing reproductive success (Moiser and Witherington, 2002; Rizkalla and Savage, 2011; Balouskus and Targett, 2012) (**Fig. 11**). 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), with the impact of artificial lighting becoming of more concern (Mazor *et al.*, 2013). For example, mortality of surf smelt embryos was 50% higher on beaches with bulkheads in Puget Sound, Washington (Rice, 2006). These types of impacts are of particular concern (Jackson *et al.*, 2008) because sea turtles are already threatened by a variety of other human

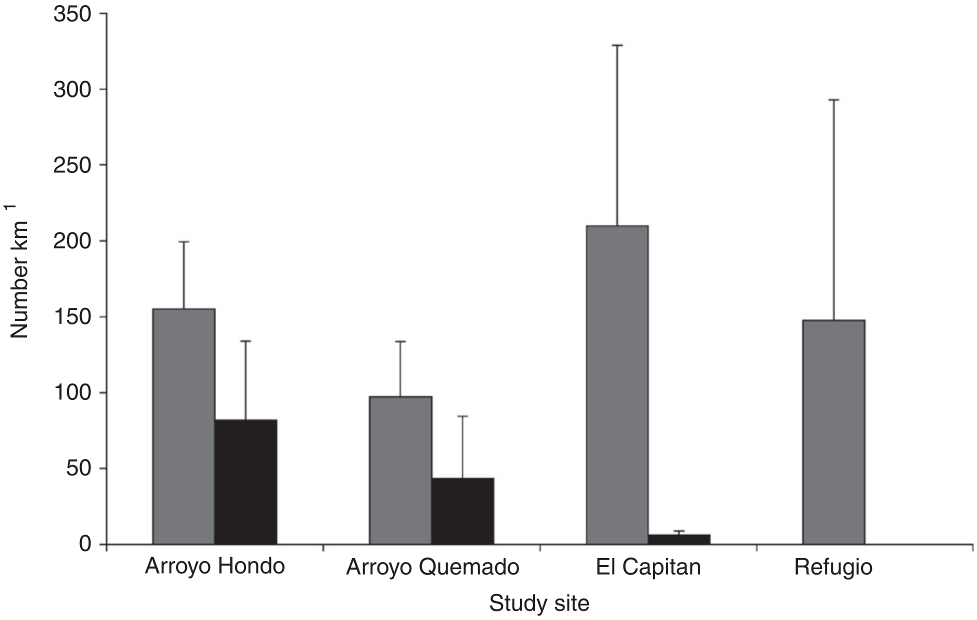

Fig. 18 Comparison of the abundance of gulls, cormorants, and herons during fall migration (Mean values + one standard error, $n = 8$) for four pairs of adjacent bluff-backed (gray bars) and armored (black bars) segments of sandy beach in August and September 2005 in California (From Dugan and Hubbard, 2006). Average abundance of these birds was 25% lower on beach segments next to armoring structures.

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).

6.2.5 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 as well as work by Perkol-Finkel *et al.*, 2006; Miller *et al.*, 2009, and Burt *et al.*, 2009). Similarly, until comparatively recently, relatively few studies of the value of armoring and urban infrastructure as habitat for marine fauna and flora were available, although many subtidal epibiota or fouling species rapidly colonize artificial structures (Glasby and Connell, 1999; Chapman and Clynick, 2006; Page *et al.*, 2006, 2008). This has changed in recent times with increasing concern about the role of novel marine habitats, such as armoring structures, and their effects on biota (see reviews by Firth *et al.*, 2014; Nordstrom, 2014; Gittman *et al.*, 2016b; Gittman and Scyphers, 2017; Strain *et al.*, 2020).

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 Airoldi, 2003; Bulleri and Airoldi, 2005; Glasby *et al.*, 2007; Landschoff *et al.*, 2013), 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

Fig. 19 A rock seawall with engineered modules installed to evaluate the responses of biodiversity to enhanced habitat complexity at different scales in Sydney Harbor Australia. Figure from Bishop, M.J., Vozzo, M,L., Mayer-Pinto, M., Dafforn, K.A., 2022 Complexity–biodiversity relationships on marine urban structures: Reintroducing habitat heterogeneity through eco-engineering. Phil. Trans. R. Soc. B 377: 20210393A.

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 sessile species (Davis *et al.*, 2002; Moreira, 2006; Sella and Perkol-Finkel, 2015). 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. (Firth *et al.*, 2016). Many of the species that have been documented as being absent from seawalls, for example, in Sydney, Australia, are species that rely on 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 (Airoldi *et al.*, 2005a; Bulleri and Chapman, 2010).

Although correlations between the structure and composition of artificial habitats and species composition have been reported (Green *et al.*, 2012), 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 experimental studies designed specifically to test such hypotheses (**Fig. 19**). In pioneering work on this concept, Chapman and Blockley (2009) demonstrated that creating artificial rock pools into a vertical seawall increased the diversity of species that colonized the wall more than threefold, both by creating shaded surfaces and by creating pools that retained water during low tide. Results of recent studies of ecological responses to increasing the structural complexity of armoring have found that effects varied with taxon, engineered microhabitat type, environmental gradients and spatial scales, but overall results suggest that increasing habitat heterogeneity of engineered structures can positively affect biodiversity (Bishop *et al.*, 2022; Clifton *et al.*, 2022; Mayer-Pinto *et al.*, 2022, also see review by Chapman *et al.*, 2018). Likewise, maritime heritage armoring structures composed of masonry blocks provided higher microhabitat complexity and supported greater intertidal biodiversity than did modern concrete structures (Baxter *et al.*, 2023).

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, although a rapidly

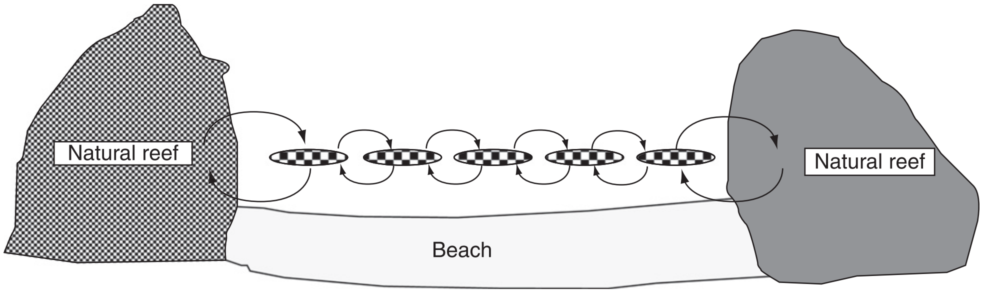

Fig. 20 Conceptual diagram of the use of coastal armoring structures as stepping stones by species that require hard substrata. The figure, not to scale, illustrates possible interactions at a regional scale of tens to hundreds of kilometers. The proliferation of armoring (indicated as) in areas with few natural rocky substrata can promote the dispersal of species outside their natural ranges, thus increasing connectedness between naturally isolated rocky reefs. Modified from Airoldi *et al.* (2005a).

expanding field (see reviews by Dafforn *et al.*, 2015; Dafforn, 2017; Chapman *et al.*, 2018; Evans *et al.*, 2019; amongst many others) 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, especially with respect to rare species, and the results, which have been obtained from relatively localized experiments, show highly variable results.

6.2.6 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 (reviewed by Bishop *et al.*, 2017). 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., Airoldi and Beck, 2007; Schlacher *et al.*, 2007; Heery *et al.*, 2017). At the same time, these structures create stepping stones or corridors for hard-bottom species (Glasby and Connell, 1999; Dethier *et al.*, 2003; Airoldi *et al.*, 2005a; Dong *et al.*, 2016), allowing the spread of species into areas where they would not occur naturally (**Fig. 20**). 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, 1990; Davis *et al.*, 2002), but all too frequently leads to the rapid expansion of documented invaders (Bulleri and Airoldi, 2005; Landschoff *et al.*, 2013). 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 Airoldi, 2003; Bulleri, 2005b; Martin *et al.*, 2005; Landschoff *et al.*, 2013), 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 km^2 of hard substrata, providing new opportunities for a variety of hard-bottom species that would be otherwise rare or absent in such sedimentary environments (Reise, 2005). Two rocky shore crab species native to the northwest Pacific coast became established and rapidly achieved high densities in artificial boulder shorelines in the Wadden Sea (Landschoff *et al.*, 2013). 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 km^2 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 fragile* spp. *tomentosoides*, the Pacific oyster *Crassostrea gigas*, the predatory snail *Rapana venosa*, and a number of tunicates) (Bacchiocchi and Airoldi, 2003; Bulleri and Airoldi, 2005; Ivešа *et al.*, 2015). 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 Airoldi, 2005; Neill *et al.*, 2006; Page *et al.*, 2006, Glasby *et al.*, 2007; Vaselli *et al.*, 2008; Dafforn *et al.*, 2009; Dafforn, 2017), and experiments clearly point to the roles of severe disturbances and sheltered conditions which are typical of habitats with extensive amounts of artificial structures (Airoldi *et al.*, 2005b; Bulleri and Airoldi, 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

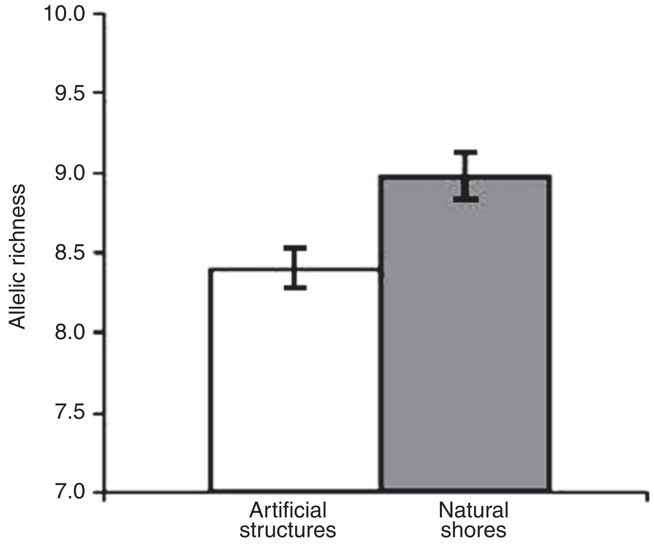

Fig. 21 Average allelic richness of the limpet, *Patella caerulea*, on artificial structures and natural shores in the Adriatic Sea based on 14 samples and 12 diploid individuals. Modified from Fauvelot, C., Bertozzi, F., Costantini, F., Airoldi, L., Abbiati, M., 2009. Lower genetic diversity in the limpet *Patella caerulea* on urban coastal structures compared to natural rocky habitats. Marine Biology 156, 2313–2323.

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 (**Fig. 21**) (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 nonindigenous 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 Airoldi, 2005; Neill *et al.*, 2006; Glasby *et al.*, 2007; Vaselli *et al.*, 2008; Dafforn *et al.*, 2009; Airoldi *et al.*, 2015). 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 Airoldi, 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 (Airoldi *et al.*, 2005b; Bulleri and Airoldi, 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.

6.2.6.1 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 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 (Airoldi *et al.*, 2005a).

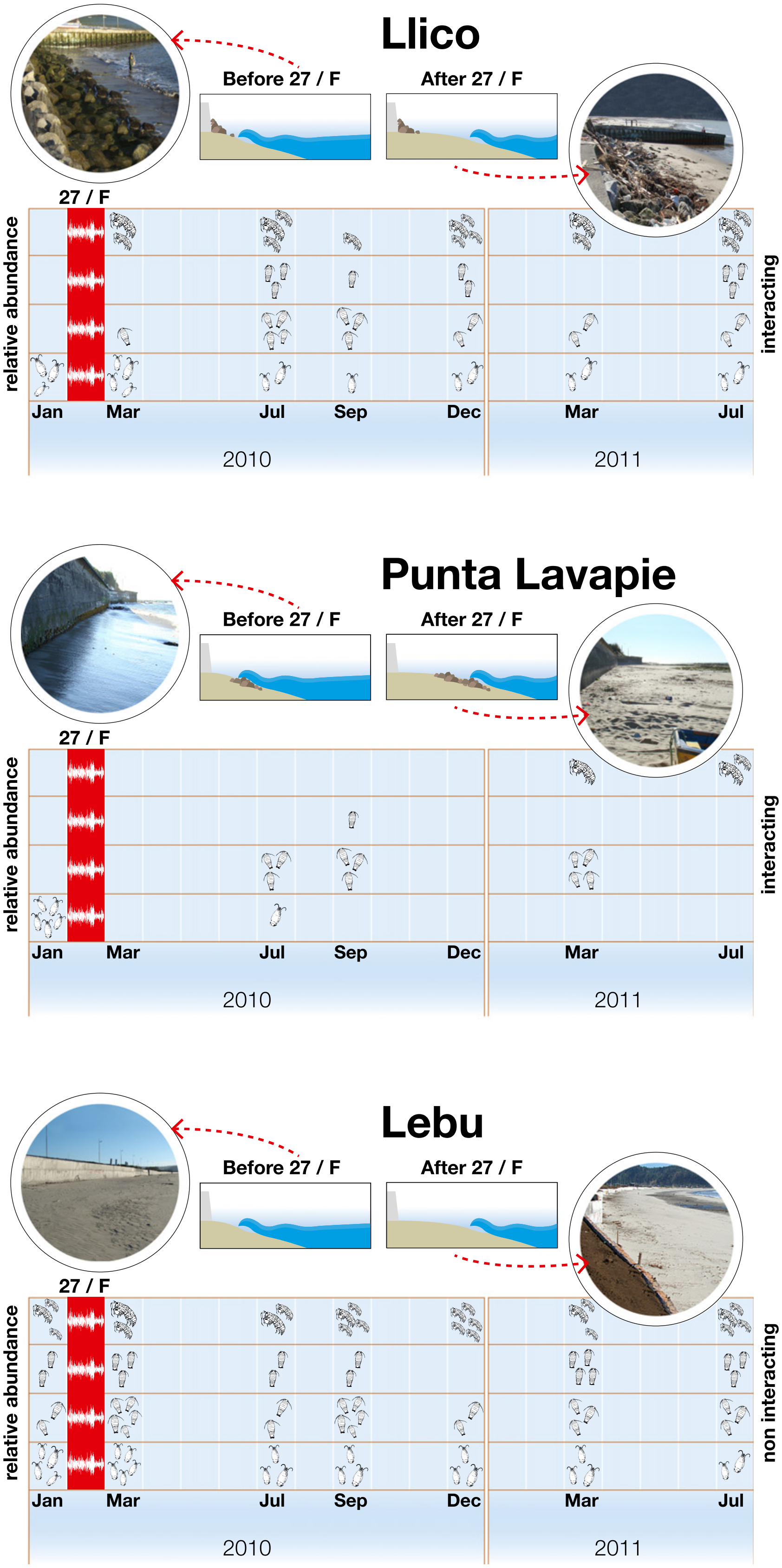

Fig. 22 Before and after images and schematics illustrating the interactions between coastal armoring, intertidal habitat and the relative abundance of mobile crustaceans that inhabit different intertidal zones for three armored sandy beaches of south - central Chile, before and after the February 2010 Maule earthquake (red columns in each illustration). The earthquake resulted in substantial coseismic uplift that expanded intertidal habitat zones. (i) Before the earthquake, waves and tides reached the rocky revetment and the seawall at Llico and Punta Lavapie (top

and middle images and plots), respectively, even during low tides and intertidal fauna were represented by only the lower intertidal anomuran crab, *Emerita analoga*. In contrast, at Lebu (bottom images and plot) no interactions between the seawall and waves and tides were generally observed during low tides and crustacean taxa of all zones were present; (ii) the width of the sandy intertidal zone at Llico, Punta Lavapie and Lebú increased after the earthquake creating new intertidal habitat. Only two weeks after the earthquake, the upper and mid beach levels of Llico, were rapidly colonized by the typical mobile crustaceans of those zones and (iii) Upper and mid intertidal crustaceans recovered more slowly at the uplifted beach of Punta Lavapie, this is probably related to the uplift of former subtidal rocky outcrops, which became intertidal after the earthquake, forming a hard rocky fringe at the lower intertidal which may have inhibited the anomuran crab, *Emerita analoga*, in the that zone. Crustaceans depicted in the graphics are arranged by their typical intertidal zone: upper intertidal talitrid amphipods (*Orchestoidea tuberculata*) in the top row, mid intertidal cirolanid isopods (*Excirolana braziliensis* and *Excirolana hirsuticauda*) in the second and third rows and low intertidal anomuran decapods (*Emerita analoga*) in the bottom row for each study site.

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 first 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.

6.2.6 Potential for Recovery/Resilience

Because of the growing concern about effects of coastal armoring, interest and effort in finding alternatives to hard armoring are increasing for both estuarine and open shorelines, for example, 'living shorelines' and nature-based solutions (Kabat *et al.*, 2009; Gittman *et al.*, 2014; Reguero *et al.*, 2018; Sutton-Grier *et al.*, 2018; Feagin *et al.*, 2021; Jordan and Fröhle, 2022). 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 (Almarshed *et al.*, 2020) and add ecological benefits (Rodriguez *et al.*, 2014; Gittman *et al.*, 2016a; Toft *et al.*, 2021). 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; Green *et al.*, 2012). In Kogarah Bay, Sydney, Australia, small patches of salt marsh have been incorporated into the shoreline armoring at mid-tidal level to provide patches of this sparse habitat, although it is not clear how much effect such small-scale additions may have on a vulnerable assemblage, and 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). An example of the potential for recovery of soft sediment intertidal habitat and biota from armoring impacts comes from comparisons made before and after the 2010 Maule earthquake ($M_W = 8.8$) and resulting coseismic uplift (up to 2.5 m) along the coast of south central Chile (Jaramillo *et al.*, 2012). For beaches where armoring structures had strongly impacted upper and mid intertidal zones and biota before the earthquake, the expansion of intertidal sandy beach habitat seaward of the structures along with the greatly reduced interaction of the armoring with waves and tides following the earthquake, allowed ecologically important mobile macroinvertebrates (talitrid amphipods and cirolanid isopods) to rapidly recolonize the upper and mid intertidal zones of some of the uplifted beaches (Jaramillo *et al.*, 2012; **Fig. 22**). Other attempts to reduce effects of past shoreline alterations and armoring or mitigate impacts of new armoring structures come under the heading of managed retreat or coastal realignment where armoring structures are removed or modified (Townend and Pethick, 2002; Morris *et al.*, 2004; French, 2008; Hino *et al.*, 2017; King *et al.*, 2018). Under this category, an approach developed to address coastal erosion for marshlands in Europe allows the sea to break through barriers and re-create fringing soft sediment 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 enhanced foraging use of restored shallow-water habitats and benthic prey 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 being constrained, it has not been universally successful (French, 2008; Hughes *et al.*, 2009) and is only appropriate where there is sufficient space available for the shoreline to shift inland and adjust dynamically to coastal processes and events.

Where coastal infrastructure is obligatory, it cannot be removed. Therefore, altering or building a structure to enhance its value as habitat, or to minimize its impact, must be the priority. Thus, coastal engineering that combines the expert knowledge of ecologists and that of engineers is needed to evaluate new ways of building infrastructure. For example, a 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; Browne and Chapman, 2014). The design of habitats such as these is necessarily a compromise between the habitats that can be added, the cost, sustainable engineering standards and what is politically acceptable (O'Shaughnessy *et al.*, 2019).

Similarly, although there are plenty of data showing correlations between the extent of infrastructure and changes to diversity of taxa, research is ongoing 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 (Iveša *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).

6.2.7 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; 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). Rising sea levels and increased destructiveness of storms may cause accelerated erosion along sedimentary shorelines and consequently inland retreat of coastlines (Slott *et al.*, 2006) with significant losses of sandy beaches (Vousdoukas *et al.*, 2020) and coastal wetlands globally (Blankespoor *et al.*, 2014) . As sedimentary shorelines retreat, society will be strongly pressed to protect infrastructure and citizens; in many cases, coastal armoring will be the engineering solution chosen 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, increased understanding of the ecological effects of coastal armoring and the feasibility of using alternatives to armoring are critical steps toward environmentally sustainable coastal management in the face of climate change.

6.2.8 Conclusions and 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 (Leo *et al.*, 2019). Coastlines are now more dynamic than ever because of changing storm patterns and sea-level rise, placing human and natural communities at greater risk (Arkema *et al.*, 2013). 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; Arkema *et al.*, 2013). 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; Bilkovic *et al.*, 2016; Guerry *et al.*, 2022), 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 and ecosystem conservation in coastal zones to preserve infrastructure, protect human communities, and preserve their livelihoods (Kareiva and Marvier, 2007). Appreciation of the benefits of ecosystem-based adaptation in marine and coastal areas and guiding principles for developing effective ecosystem-based adaptation strategies are growing (Hale *et al.*, 2009; Airoldi *et al.*, 2021; Arkema *et al.*, 2013; Sutton-Grier *et al.*, 2018;

Schroder *et al.*, 2022). 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 into the future.

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