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MINI-REVIEW

The introduction of coastal infrastructure as a driver of change in marine environments

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Summary

1. Coastal landscapes are being transformed as a consequence of the increasing demand for urban infrastructure to sustain commercial, residential and tourist activities. A variety of man-made structures, such as breakwaters, jetties and seawalls have thus become ubiquitous features of intertidal and shallow subtidal habitats. This transformation will accelerate in response to the exponential growth of human populations and to global changes, such as sea-level rise and increased frequency of extreme meteorological events (e.g. storms). Here, we provide a critical overview of the major ecological effects of increasing infrastructure to marine habitats, we identify future research directions for advancing our understanding of marine urban ecosystems and we highlight how alternative management options might mitigate their impacts.

2. Urban infrastructure supports different epibiota and associated assemblages and does not function as surrogate of natural rocky habitats. Its introduction in the intertidal zone or in nearshore waters can cause fragmentation and loss of natural habitats. Furthermore, the provision of novel habitat (hard substrata) along sedimentary shores can alter local and regional biodiversity by modifying natural patterns of dispersal of species, or by facilitating the establishment and spread of exotic species.

3. Attempts to use ecological criteria to solve problems of urban infrastructure are promising. Incorporating natural elements of habitat (e.g. wetland vegetation; seagrass) into shoreline stabilization can reduce ecological impacts, without impinging on its efficacy in halting erosion. Likewise, improving the ecological value of artificial structures by adding features of habitat that are generally missing from such structures (e.g. rock-pools) can contribute to mitigation of the detrimental effects of urbanization on biodiversity. Management of anthropogenic disturbances (e.g. maintenance works; harvesting) to artificial habitat is, however, necessary if such attempts are to be successful.

4. *Synthesis and applications*. Increasing our understanding of the ecological functioning of marine habitats created by urban infrastructure and incorporating ecological criteria into coastal engineering are crucial for preserving biodiversity in the face of the growth of human populations in coastal areas and of forecasted global changes. Achieving this goal will need strong collaboration between engineers, managers and ecologists.

Key-words: artificial habitats, coastal engineering, coastal erosion, exotic species, global climate change, man-made structures, marine biodiversity, urban infrastructure, urbanization

Introduction

The human population on Earth, which was estimated at 6 billion in 1999, is projected to increase by 50% in less than 50 years (U.S. Census Bureau 2009). Many of the largest cities in the world are located in coastal zones (Timmerman & White 1997) and more than 75% of people are expected to live within 100 km of a coast by 2025 – a world-wide phenomenon (EEA 2006). In 2003, 53% of the population of the United States lived in the 673 coastal counties and this is expected to increase

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by more than 12 million people by 2015 (NOAA 2004). Coastal areas are also over-burdened by mass tourism. The Mediterranean coast may have 350 million seasonal tourists per year by 2025 (Hinrichsen 1999). The transformation of the coastal town of Cancun in Mexico into an international tourist destination has resulted in the resident population of about 300 000 being swamped annually by 2 million visitors (Burke *et al.* 2001). These major changes to the demography and distribution of human populations has severely impacted coastal landscapes, which are continually being altered by the addition of the infrastructure needed to sustain residential, commercial and tourist activities. For example, in several regions of Italy, France and Spain, built-up areas in the coastal strip exceed 45% of land-cover (EEA 2006). Transformation of coastal landscapes in response to urbanization is not, however, limited to the land because the intertidal zone and nearshore estuarine and marine waters are also increasingly altered by the loss and fragmentation of natural habitats and by the proliferation of a variety of built structures, such as breakwaters, seawalls, jetties and pilings (Table 1).

Armouring shorelines, by means of riprap revetments, seawalls or bulkheads, is a very common approach to combating erosion, especially in urban areas where there is great demand for property near the shore (Davis, Levin & Walther 2002; Living Shoreline Summit Steering Committee 2006; Airoldi & Beck 2007). A considerable proportion of the U.S. coastline is subject to erosion and has been 'hardened' to protect against damage to infrastructure, with more than 50% of some coastlines in California, Virginia or Maryland having replaced natural soft habitats with artificial hard surfaces (Living Shoreline Summit Steering Committee 2006). The use of hard coastal-defence structures is also predicted to increase in response to forecast sea-level rise and increased intensity and frequency of large storms (Michener *et al.* 1997; Thompson, Crowe & Hawkins 2002). Artificial structures are therefore becoming ubiquitous features of coastal waters in urbanized centres where they can form the dominant intertidal and shallow subtidal habitat (Russell *et al.* 1983; Davis *et al.* 2002; Chapman & Bulleri 2003; Airoldi *et al.* 2005a; Living Shoreline Summit Steering Committee 2006; Airoldi & Beck 2007).

Despite these widespread changes, ecological issues related to the introduction of infrastructure into shallow coastal waters have received relatively little attention (Southward & Orton 1954; Glasby & Connell 1999; Hawkins *et al.* 2002; Chapman 2003; Bulleri 2006). For instance, a recent review of human impacts on marine ecosystems (Halpern *et al.* 2008) did not include the proliferation of artificial structures among key anthropogenic drivers of ecological change to coastal habitats, possibly because of the lack of relevant information. Although some reviews have acknowledged the impact of development on loss of coastal habitats (e.g. Thompson *et al.* 2002; Airoldi & Beck 2007; Branch *et al.* 2008), relatively few

Table 1. Purposes and characteristics of common urban infrastructures deployed in near-shore waters

Type of structure	Action and purposes	Materials used	Positioning/ orientation respect to the shore	Position respect to the sea surface	Wave exposure
Breakwaters	Reduce the intensity of wave forces in inshore waters; used for protecting ports, harbours and marinas and as coastal defences	Sandstone; geo-textile; granite; sandbags; concrete; wood	Not connected to shore parallel or fish tail	Emergent; low crested; submerged	Exposed
Groynes	Reduce along-shore transport of sediments; used in coastal defence schemes, often in association with breakwaters	Sandstone; geo-textile; granite; concrete; wood; sandbags	Connected to shore perpendicular	Emergent; low crested; submerged	Exposed
Jetties	Reduce wave- and tide-generated currents; used for developing, ports, harbours, marinas and as constituents of coastal defence schemes	Sandstone; geo-textile; granite; concrete; wood; sandbags	Connected to shore perpendicular	Emergent; low crested; submerged	Exposed
Seawalls Bulkheads	Reduce the impact of waves on shore; used as a tool against coastal erosion and as a constituent of ports, docks and marinas	Sandstone; geo-textile; granite; concrete; steel; vynil; sandbags; wood	Onshore parallel on open coasts, but variable in enclosed waters	Emergent	Exposed to sheltered
Pilings	Sustain infrastructure, such as bridges, piers, docks and for the mooring of vessels	Concrete; wood; fibreglass; metal	Onshore to offshore	Emergent	Exposed to sheltered
Floating docks	Create boating facilities	Concrete; wood; plastic fibreglass; metal	Connected to shore varying orientation	Emergent	Sheltered
Ropes-poles cages-nets	Constituents of aquaculture facilities	Fabric; plastic; wood; fibreglass; metal	Not connected to shore varying orientation	Emergent; submerged	Moderately exposed to sheltered

studies have explicitly addressed ecological impacts of local, but extensive, deployment of infrastructure (Glasby & Connell 1999), nor assessed any ecological value that such habitats may provide in urban environments (Moschella *et al.* 2005; Chapman 2006; Clynick, Chapman & Underwood 2008). It is likely that the lack of interest in ecological features of urban infrastructure lies in the fact that, in contrast to artificial reefs (Baine 2001), it is not introduced with the primary objective of enhancing populations of particular taxa or to mitigate the impacts of human activities (Raymundo *et al.* 2007), so that changes eventually caused to biota are often considered as a side effect (Bulleri 2005a).

Our objectives are to: (i) summarize current knowledge of the ecological impacts of urban infrastructure on marine habitats; (ii) discuss the implications of alternative coastal defence options; and (iii) identify future research directions that might advance understanding of these artificial habitats and, hence, our ability to preserve biodiversity in urbanized environments. Due to substantial differences in their objectives, literature on artificial reefs deserves to be reviewed on its own (e.g. Bohnsack & Sutherland 1985; Baine 2001) and will be not explicitly included in this review, which focuses solely on the side effects of urban infrastructure.

Ecological consequences of the introduction of artificial structures

INTRODUCTION OF NOVEL HABITAT

The ecological impacts of coastal infrastructure in shallow coastal waters vary according to the nature of the surrounding habitat(s) (Bulleri 2005a). Introducing artificial surfaces onto rocky bottoms is sometimes considered not to alter the fundamental nature of the habitat, especially when these structures are built with natural stone. It has, in fact, been assumed that the structure and functioning of assemblages that colonize those surfaces are analogous to those living on adjacent natural rocky shores (Southward & Orton 1954; Hawkins, Southward & Barrett 1983; Thompson *et al.* 2002; Branch *et al.* 2008). There is, however, mounting evidence that epibiota living on and fish assemblages associated with artificial structures, such as breakwaters, seawalls, groynes or floating pontoons, differ from those on natural reefs (Lincoln-Smith, Hair & Bell 1994; Glasby & Connell 1999; Rilov & Benayahu 2000; Perkol-Finkel & Benayahu 2004; Bulleri, Chapman & Underwood 2005; Moschella *et al.* 2005; Clynick, Chapman & Underwood 2008; Lam, Huang & Chan 2009).

Coastal infrastructure differs from natural habitats in many ways, each of which can affect which species can establish populations on it. The different types of infrastructure generally provide vertical habitat, whereas many natural habitats slope more gently or have heterogeneous topography (Chapman 2003; Perkol-Finkel & Benayahu 2004; Moschella *et al.* 2005; Lam *et al.* 2009). Many species of intertidal or subtidal animals and plants are limited in their distribution by the slope of the substratum (Whorff, Whorff & Sweet 1995). In addition, artificial intertidal vertical surfaces may severely crowd species into a limited area compared with the amount of intertidal habitat available on more gently sloping natural shores. Thus, densities may be abnormally increased, or species that do not usually come into contact can be forced to occupy the same area, potentially increasing the strength of interspecific interactions. For example, abnormally large densities of an intertidal limpet on seawalls in Australia resulted in smaller adult size and reduced reproductive output (Moreira, Chapman & Underwood 2006).

Artificial structures are also often constructed of unnatural material, (e.g. concrete, plastic or metal; Attrill *et al.* 1999) which may affect colonization, even if many epibiota readily colonize unnatural surfaces (Svane & Petersen 2001; Chapman & Clynick 2006). For example, fish and epibenthic assemblages that developed on a submerged breakwater made of sand-filled geo-textile containers resulted in very different assemblages from those found on nearby rocky reefs, probably as a consequence of the physical properties of the substratum affecting recruitment (Edwards & Smith 2005). Again, mussels *Mytilus trossulus* growing on shellfish farming structures grew larger, had lower shell mass and weaker byssal attachment in comparison with those on adjacent intertidal areas, ultimately altering the feeding behaviour and local distribution of wintering sea ducks (Kirk, Esler & Boyd 2007). In contrast, Glasby (1999) showed that differences between assemblages on natural reefs and wooden pilings were caused by shading of the pilings and proximity of the different habitats to the seafloor, rather than by the features of the structures themselves.

Larvae might not perceive differences between artificial structures and rocky shores at small scales, such as those influencing their settlement, especially when the former are made of the same material as the natural shore. Nevertheless, the lack of appropriate microhabitats (e.g. rock-pools, over-hangs) that function as refuges against predators or stressful environmental conditions (e.g. wave action or desiccation in intertidal habitats), in comparison with natural hard surfaces, could influence post-settlement survival of plant or animal propagules (Bulleri 2005b; Moschella *et al.* 2005). Also, many built structures are deployed on sedimentary bottoms and severe sand-scouring can be an important cause of mortality for sessile organisms, ultimately leading to relatively low levels of species richness (Hawkins *et al.* 1983; Moschella *et al.* 2005).

Built structures often provide unnatural sheltered habitats along wave-exposed coasts. Seawalls, pontoons and pilings which are enclosed in marinas and the landward sides of breakwaters running parallel the shore create very sheltered conditions. The reduced water flow, turbidity or abrasion by sediments in these novel sheltered habitats can promote the establishment of assemblages that differ in species richness, composition or relative abundances from those associated with nearby natural exposed rocky habitats (Bulleri & Chapman 2004; Clynick 2007; Vaselli, Bulleri & Benedetti-Cecchi 2008).

The age of introduced surfaces is also an important determinant of the extent to which the assemblages they support can resemble those on natural rocky substrata (Karlson 1978; Hawkins *et al.* 1983; Pinn, Mitchell & Corkill 2005; Moschella *et al.* 2005). Assemblages on artificial structures that have been in place for several decades may be more similar to those on adjacent rocky shores (Pister 2009), perhaps as a result of increased heterogeneity of the substratum because of weathering (e.g. Plymouth Breakwater since 1830s; Southward & Orton 1954). In other cases, however, it is clear that artificial structures such as seawalls do not support natural assemblages, even after many decades (Chapman 2003).

Finally, the mobility of some built structures, such as buoys and floating pontoons, affect the structure of epibiota (Connell 2000; Perkol-Finkel *et al.* 2008; Dafforn, Johnston & Glasby 2009) and underlie the development of different assemblages from those occurring on natural hard substrata (Glasby 2001; Holloway & Connell 2002).

Thus, key ecological processes, such as recruitment (Glasby 1999; Bulleri 2005b), foraging (Bulleri, Chapman & Underwood 2004), competition, predation (Kirk *et al.* 2007) or reproduction (Moreira *et al.* 2006) may differ between natural and artificial habitats and current data indicate that artificial structures do not function as surrogates of natural habitat. Very little research has, however, been done to identify the mechanisms that cause differences in patterns or processes between natural and artificial rocky habitats.

EFFECTS ON ADJACENT HABITATS

There has been little focus on the effects of urban infrastructure on adjacent rocky habitats, although Goodsell, Chapman & Underwood (2007) showed differences in diversity of species living on intertidal rocky shores that were bordered by seawalls compared with shores bordered by natural habitat. Shores bordered by seawalls were typically smaller and spaced further apart, which suggests that differences in the biota may be associated with fragmentation of habitat, but what precisely caused the differences is not known. The biota living on artificial structures may also provide an important food-source for species living in adjacent waters (Caine 1987).

Shading of the substratum can, however, be a major impact of introduced infrastructure, reducing the cover of plants in saltmarsh (Sanger, Holland & Gainey 2004) and diversity of fish around docks (Able, Manderson & Studholme 1998). In contrast, shading by wharves in Sydney Harbour (Australia) increased the diversity of species living on seawalls themselves (Blockley 2007) or within constructed microhabitats on these seawalls (Chapman & Blockley 2009), reducing, to some extent, the negative impact that the walls themselves had on intertidal diversity (Chapman 2003). Also, sheltering of natural rocky shores by infrastructure (i.e. breakwaters) has been shown to cause a shift from assemblages dominated by consumers to those dominated by primary producers, implying a substantial alteration in the functioning of the system (Martins *et al.* 2009).

Building structures over or onto soft sediments has multiple ecological implications (Bulleri 2005a). There is considerable evidence that the deployment of coastal defence structures, such as breakwaters and groynes, changes water flow, illumination and rates of sedimentation (Bertasi *et al.* 2007), all of which can have detrimental effects on organisms in sediments (e.g. Davis, Van Blaricom & Dayton 1982; Lindegarth 2001; Airoldi *et al.* 2005a; Martin *et al.* 2005). Shoreline armouring can result in increased steepness of the shore and deeper near-shore waters and, ultimately, in profound alterations to intertidal and shallow-water soft-bottom habitats (Roberts 2006; Ocean Studies Board 2007). Furthermore, seawalls or bulkheads placed at the landward edge of soft-sedimentary habitats (e.g. beaches, saltmarsh or mangrove wetlands) will prevent inland migration of these habitats if sea-level rises, potentially causing loss of nursery and foraging grounds for fish and shorebirds, or breeding sites for turtles (Hulme 2005; Gilman, Ellison & Coleman 2007; Schleupner 2008).

REGIONAL OR LARGE-SCALE CHANGES

Disruption of water flow by infrastructure may stop or limit dispersal of propagules, which may, in turn, alter the connectivity within metapopulations, changing the relative proportions of source and sink populations. Connectivity among marine populations is generally maintained by movement of larvae and propagules (Kinlan & Gaines 2003) – an ‘invisible ‘process – and there is little current knowledge about how connected are marine populations. Recent research suggests, however, that populations of many species are not as open as once thought and the occupation of large areas by a species may be maintained by relatively few breeding adults (Swearer *et al.* 1999). For example, disruption of along-shore currents by groynes (Burcharth *et al.* 2007) could cause the accumulation of larvae and propagules at particular sites on the coast. If this phenomenon is widespread, then local disruptions to water flow, if in crucial areas, may have large effects on sink populations elsewhere.

Irrespective of whether offshore structures (breakwaters) or direct hardening of the shoreline are used to combat erosion, the use of hard substrata can also attract species that usually live in association with rocky reefs into areas where they may have no natural habitat. Although this could be viewed as enhancement of local species richness, this opinion is dependent on the local context and enhanced populations are not necessarily desirable. There is some evidence suggesting that artificial structures may create novel habitat in mitigation for that lost, or may enhance populations of fish (Barwick *et al.* 2004; Guidetti, Bussotti & Boero 2005) or endangered invertebrates (Guerra-Garcia *et al.* 2004; Wen *et al.* 2007). Indeed, the use of artificial structures has been advocated as a potential tool to facilitate the poleward migration of species in response to climate changes, such as sea-water warming (Hoegh-Guldberg *et al.* 2008). On the other hand, urbanization can contribute to ongoing homogenization of biota at a global scale, especially if it supports large populations of a similar and reduced set of species (McKinney & Lockwood 1999; Branch *et al.* 2008). Built structures can function as corridors or stepping stones (Glasby & Connell 1999), connecting otherwise separated populations. For example, Sammarco, Atchison & Boland (2004) found that oil and gas platforms enhanced the dispersal of coral populations in the Gulf of Mexico, including dispersal into areas where they were previously absent. Increased dispersal along artificial corridors can also facilitate gene flow, ultimately reducing genetic diversity (Airoldi *et al.* 2005a). Thus, proliferation of man-made structures along the sandy Belgian coast has not only enabled the dispersal of the periwinkle *Littorina saxatilis*, which lack a planktonic larval stage, but also reduced the genetic variability of populations compared to those on natural shores (Johannesson & Warmoes 1990). Biotic homogenization is currently seen as a major anthropogenic impact (Sax & Gaines 2003), but, despite important evolutionary consequences, the potential role of marine artificial structures in promoting gene exchange remains virtually unexplored (but see Fauvelot *et al.* 2009).

Currently, poor understanding about which species do or do not use artificial habitat provided by urban structures means that the characteristics of the assemblages that are likely to establish on or near these structures are not predictable in advance.

The assessment of any ecological impacts of coastal infrastructure must be evaluated against potential benefits for end-users. There are the direct anthropogenic benefits of the infrastructure itself, which are determined without any consideration of the ecological value of the structure. Although, in many countries, there are now increasing limits on the spread of urbanization without due consideration of 'green space' (Sandström, Angelstam & Khakee 2006), this is generally not the case for the spread of urbanization in coastal waters. There are also indirect effects, e.g. changes in water quality because of urban runoff, poor circulation (Morton 1992) or the introduction of new species. These may be considered a nuisance (e.g. the spread of ephemeral algae), or may have positive effects (e.g. enhanced populations of fish). Although both of these taxa are equally 'exotic', one may be preferred over the other because of economic reasons. Thus, it may be deemed necessary to spend money removing algae because of public perception, but increased populations of fish may be perceived as an added 'bonus' to an area. Similarly, biofiltration of water by fouling assemblages could be viewed as a benefit generated by the deployment of infrastructure (Hughes, Cook & Sayer 2005), despite potential alterations to local biodiversity.

The use of increased shoreline armouring cannot be avoided when coastal infrastructure and property are at risk, but options for mitigating their ecological impacts should be given adequate attention during planning. When specific characteristics of the biota that may be supported by any infrastructure (e.g. the attraction or enhancement of target species) are defined and accepted in advance of building works, for example, as in some programmes of restoration (e.g. Iverson & Bannerot 1984), local or regional enhancement of biodiversity may be considered as a positive effect of urbanization.

IMPLICATIONS FOR THE SPREAD OF EXOTIC SPECIES

Artificial structures seem to be particularly susceptible to invasion (Bulleri & Airoldi 2005; Glasby *et al.* 2006; Neves *et al.* 2007; Tyrrell & Byers 2007). This may be simply because of the fact that they are found mostly in areas that are characterized by poor environmental conditions, frequent disturbances (e.g. maintenance works), or those that support activities linked to importation of exotic species (e.g. shipping, aquaculture). Thus, the structures themselves may be a minor factor in invasions, although providing hard substrata in areas that do not naturally have rocky reefs can exacerbate spread of species. The spread of the green introduced macroalgae, *Codium fragile* ssp. *tomentosoides* (Bulleri & Airoldi 2005) and *Caulerpa racemosa* (Vaselli *et al.* 2008) along the coasts of Italy has been fostered by the presence of breakwaters on which they could establish viable populations.

There is also evidence that many exotic species actually establish populations more easily on artificial structures and that there may be a greater proportion of exotic to native species on artificial compared to adjacent natural habitat (Glasby *et al.* 2006). Vulnerability of artificial structures to invasion might be due to their physical attributes or to indirect effects of other species living on them. The shoreward side of breakwaters, for example, creates wave-sheltered environments, which are particularly prone to invasion by some green macroalgae (Bulleri & Airoldi 2005; Vaselli *et al.* 2008). Colonization by exotic epifauna may be enhanced on shallow moving substrata, such as floating docks (Dafforn, Johnston & Glasby 2009). Filter-feeding invaders which are transported on ship hulls are generally well adapted to resist high shear stress and could readily take advantage of available space on moving artificial surfaces (Dafforn, Johnston & Glasby 2009).

On the other hand, assemblages on artificial structures are often characterized by a smaller diversity of species compared with natural habitats (Bacchiocchi & Airoldi 2003; Chapman 2003; Martin *et al.* 2005) and the establishment of invaders could be therefore enhanced by weaker competitive interactions with extant species, or by smaller mortality because of predation, as postulated by the biotic resistance (Elton 1958) and enemy release hypotheses (Keane & Crawley 2002), respectively. Understanding the factors and processes sustaining biodiversity in the artificial habitats created by urban environments and assessing their influences on the ability of exotic species to get established is therefore a key to enhance our ability to predict and manage future pathways of invasion in coastal areas.

Alternative management options

Recently, there has been considerable effort to identify solutions to coastal erosion using alternatives to hard armouring (e.g. 'Living Shorelines'; Living Shoreline Summit Steering Committee 2006). For example, incorporating natural elements of habitat, such as wetland vegetation, seagrasses, coarse woody debris, or shellfish reefs into shoreline stabilization can reduce ecological impacts without impinging on their efficacy in halting erosion. Thus, natural habitats alone may provide a buffer against erosion in sheltered areas, but a combination of natural habitat with hard structures ('hybrid' designs) might be necessary in higher wave energy environments (Smith 2006). Both approaches provide clear ecological advantages over traditional armouring of the shore, such as increased primary productivity, improved water quality or enhancement of habitats for birds, amphibians and crabs (Subramanian *et al.* 2006).

Managed retreat or realignment of hard coastal defence structures (mostly seawalls) has been identified as an adaptive strategy for alleviating estuarine flood risk or for the re-establishment of ecologically valuable intertidal habitats, such as saltmarshes and tidal flats (Townend & Pethick 2002; Morris *et al.* 2004; French 2008). This involves dismantling or breaching shore defences and eliminating them or moving them inland, preferably taking advantage of natural topographic contours to reduce the cost of engineering to the standards required for alleviating flooding risks (French 2008; Townend 2008). Although this alternative to further armouring of shorelines is promising, there are concerns for its long-term performance (French 2008; Hughes, Fletcher & Hardy 2009) and it is only feasible where there is the possibility of the shoreline being relocated inland. Subsidence of sites caused by shoreline armouring has, in some cases, prevented the achievement of restoration targets (Hulme 2005). There is also evidence for assemblages in re-established tidal habitats to differ in terms of species richness and composition compared with natural sites, as documented by Garbutt & Wolters (2008) for saltmarshes in the Essex estuary (UK), and for intense damage from erosion (Hughes *et al.* 2009).

In many cases, infrastructure is, however, obligatory, either for public safety or to meet engineering standards (e.g. ports, roads, bridges or wharves). Under these circumstances, minimizing their ecological impacts should be considered a priority. Although there has been research on using shoreline armouring to assist in restoration of terrestrial habitat (e.g. coastal dunes; Comoss, Kelly & Leslie 2002), or for creating habitat for fish (Baine 2001), there has been relatively little effort to improve designs of these structures in marine settings to achieve secondary management end-points (Mann 1988). These include the provision of suitable habitat to counteract deleterious effects of urbanization for species living on or adjacent to these structures, to enhance living resources exploitable as food (fish and shellfish) or to promote recreational (e.g. birdwatching, snorkeling) and educational activities (Zedler & Leach 1998; Burcharth *et al.* 2007). For example, Burt *et al.* (2009) have shown that the use of Gabbro as a material to construct breakwaters could encourage the recovery of corals in tropical regions, while Russell *et al.* (1983) have shown that disused docks can support diverse assemblages of marine animals and plants and can be used to promote educational, amenity and economic activities.

The future of sustaining biodiversity in urbanized areas cannot rely on simply documenting the effects of urban development on biodiversity, but urgently needs better collaboration among engineers, managers and ecologists to develop improved ways of building infrastructure to provide habitat for more species without compromising engineering standards. Collaborations between engineers and ecologists, taking advantage of the principles of experimental design that have advanced ecological understanding in natural habitats, can enable tests of hypotheses about different ways of building new infrastructure, or changing the face of very altered, urbanized environments (Chapman & Clynick 2006; Chapman & Blockley 2009).

For example, recent research in Australia has focused on adding novel intertidal habitats into flat, featureless façades of seawalls in a cost-effective manner that neither compromises safety nor other engineering requirements, but increases diversity of species living on the wall itself. Certain habitats were designed to resemble intertidal rock-pools (Chapman & Blockley 2009), but, of course, they could not completely mimic natural pools as they were placed in a vertical wall (Fig. 1a, b). The design of these habitats was therefore a compromise between engineering constraints and the requirements for habitats that retained water during low tide. Their construction was entirely under the control of engineers, but where and how they were placed was determined by the requirements for multiple sites and adequate replication at different tidal heights, so that the data could be used to test specific ecological hypotheses. These pools increased the diversity of species of algae and sessile animals many-fold, especially higher on the shore where environmental conditions are harsh (Fig. 2). Although preli-

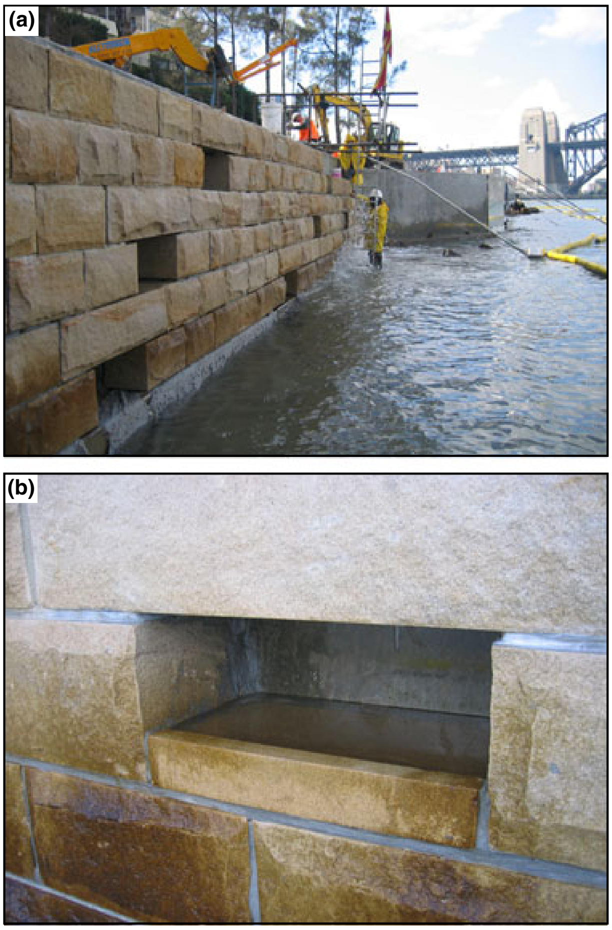

Fig. 1. (a) Intertidal 'rock-pools' constructed in the vertical face of a seawall in Sydney Harbour (Australia). These features of habitat were introduced to seawalls to mitigate effects of loss or degradation of rocky platforms on intertidal biodiversity. (b) Details of a rock-pool retaining water during low tide.

minary, these results point at the importance of this sort of research, especially as seawalls will proliferate with both increasing urbanization and sea-level rise.

Similarly, results from the EU project DELOS suggest that detrimental effects of the reduction of water flow on organisms living in sediments caused by the deployment of coastal defence structures that run parallel to the coastline can, at least, be partially mitigated if relevant ecological criteria are taken into account during the planning of the intervention (Airoldi *et al.* 2005a; Martin *et al.* 2005; Burcharth 2007). In fact, low-crested barriers allowing greater water exchange between the protected landward side and the open sea, in comparison with emerged barriers, would reduce the deposition of fine sediments and the accumulation of organic materials (e.g. algal or seagrass debris), with a consequent improvement of physical and chemical conditions in sediments. In practice, designs attempting to maximize the over-topping by waves and the porosity of breakwaters, while minimizing their length and avoiding the enclosure of the protected zone by means of lateral groynes, might lead to a valid compromise between the need to achieve the primary objective for which the intervention is planned (halt erosion) and the protection of local biodiversity (Burcharth *et al.* 2007). Further technical aspects that can promote the diversity of epibenthic assemblages include the construction of a berm around the structures to minimize sand-scouring and the use of materials that are more easily weathered or bioeroded, leading to more complex and heterogeneous surfaces (Burcharth *et al.* 2007).

A further consideration concerns how such structures are maintained or repaired. For example, crevices between blocks of stone or concrete on a wall can provide useful habitat if the mortar is not made flush with the surface of the wall (M. G. Chapman, unpublished data). Seawalls are, however, often repaired purely for aesthetic reasons and this usually involves filling these gaps to provide a flat featureless façade. Disturbance caused by maintenance works associated with the use of artificial structures can reduce the diversity of epibiota because colonization is repeatedly reset by the disturbance. This can lead to continued dominance by early opportunistic species, such as ephemeral filamentous algae (Bacchiocchi & Airoldi 2003; Burcharth *et al.* 2007). Likewise, frequent disturbances from recreation in urban centres, such as harvesting, can change the structure of assemblages on artificial habitats. Thus, recreational harvesting of mussels on breakwaters in the northern Adriatic Sea creates unoccupied space, which favours colonization by native and exotic macro-algae (Airoldi *et al.* 2005b). Therefore, attempts to use ecological criteria to solve problems of urban infrastructure must be framed within the context of continued management of the habitat and other anthropogenic disturbances if it is to be successful.

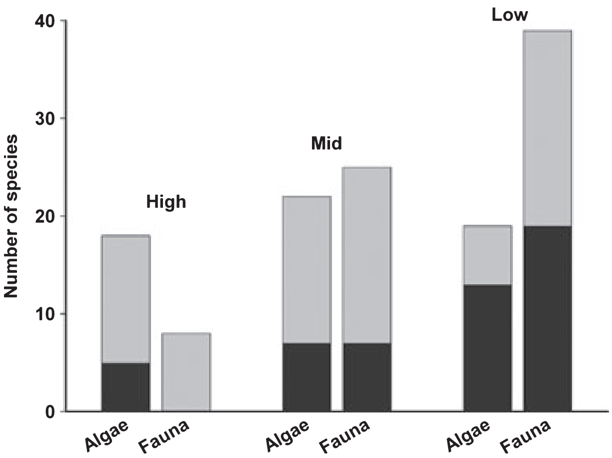

Fig. 2. The number of species of macro-algae and sessile animals living on the façade of the seawall (black bars) and the number of additional species found in the 'rock-pools' (clear bars) at three different shore levels (high, mid and low); data summed across all sites (see Chapman & Blockley 2009 for details of the experimental design and analyses).

Future directions for research

Although data are limited, there is clear evidence that urban infrastructure has adverse effects on extant natural intertidal and shallow subtidal marine habitats (including soft- or rocky bottoms) and does not generally support natural intertidal or subtidal assemblages. It provides unsuitable or sub-optimal habitat for many species and favours the establishment and spread of introduced and invasive species that rapidly exploit disturbed and new habitat. Research must now progress beyond documenting spatial patterns of distribution and abundance in these altered environments towards understanding how fundamental ecological processes (competition, predation, facilitation, etc.) are affected because, ultimately, the sustainability of biota in highly altered habitats is dependent on the sustainability of natural ecological interactions.

These ecological interactions need to be investigated on artificial structures in urban settings, where the characteristics of the habitat itself as well as the suite of species occupying it differ from normal and where there are continuous additional disturbances. Studies will need to incorporate rigorously designed field experiments with adequate replication and at spatial and temporal scales relevant to managers so that the results of the experiments can underpin future management practices.

Thus, a second important direction for research involves ecological (in contrast to ecosystem) engineering (Schulze 1996). This melds engineering theory and practice with ecological understanding, particularly with regards to uncertainty of ecological processes. If artificial structures are to be built to reduce changes to natural assemblages and to create improved habitats in addition to their primary role, it is important that engineering practices become flexible and recognize that the same design may have different outcomes in different places. It is not yet possible to provide a 'recipe book' of ecological engineering, but with more experimental collaborations between engineers and ecologists, progress will be made.

Concluding remarks

Urban environments can be considered novel or emerging ecosystems (Hobbs *et al.* 2006), with changed biotic (e.g.

homogenization of species) and abiotic conditions (e.g. altered nutrients, higher temperatures, novel habitats), most of which are probably irreversible. The importance of these novel conditions for sustaining biodiversity has received some attention from terrestrial ecologists (Miller & Hobbs 2002), but little from marine ecologists (Bulleri 2006) and most ecological understanding of marine environments has come from research in relatively unaltered areas. Successful conservation or management of species needs robust and up-to-date ecological knowledge and theories. We need to understand how much of current theory is applicable to coastal areas where infrastructure has destroyed and fragmented habitats, potentially disrupted connectivity among populations and altered the mosaic of patches of habitats at the scale of the seascape. Achieving these goals should be given high priority in research agendas of both funding bodies and marine ecologists if we are to develop the tools necessary to face the progressive increase in the use of man-made structures that is being triggered by the spread of human populations and by the need to take action against consequences of climate change.

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