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REVIEW

Coastal ecological engineering and habitat restoration: incorporating biologically diverse boulder habitat

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ABSTRACT: Ecological engineering is increasingly being studied and applied in order to reverse declines of biological diversity caused by coastal urbanisation and habitat degradation. As methods become more sophisticated and the theoretical framework more advanced, engineering of more complex and biologically diverse habitat types becomes possible. This review discusses the benefits of incorporating boulder habitat, which provides a unique combination of intermediate stability and high structural complexity, and can be occupied by many rare species. The inclusion of this habitat into engineered coastlines would therefore represent an important outcome for coastal ecological engineering by providing habitat for these species. Some methods are already in use to restore degraded boulder habitat; these methods should strive to closely mimic boulder habitat because semi-natural habitats (e.g. building rubble at bases of seawalls) have not been found to support rare species at this stage. Creation of new boulder habitat is also valuable for important fisheries (e.g. *Haliotis* spp.). Methods will be improved by focusing on small-scale microhabitats created by boulders and how these microhabitats provide shelter from locally relevant predators. Boulder habitat can reliably stabilise shorelines whereas alternative ecological engineering options based on littoral vegetation (e.g. mangroves, seagrass or saltmarsh) provide stabilisation involving strong spatiotemporal variability. Ecological engineering methods that include highly novel habitats, such as boulders, will achieve valuable biodiversity outcomes by allowing large-scale increases in along-shore distribution of specialist species. Overall, incorporation of boulder habitat in ecological engineering will help ensure coastal habitats include highly diverse assemblages and important ecological functionality as the pressure to modify coastlines increases.

KEY WORDS: Coastal urbanisation · Habitat complexity · Boulder reefs · Sea ranching · Seawalls · Fisheries

INTRODUCTION

Marine conservation (Abelson 2006), fisheries (Baine 2001, Dixon et al. 2006), and sustainable development of coastal infrastructure (Chapman & Underwood 2011, Perkins et al. 2015) increasingly rely on habitat construction/restoration and ecological engineering to achieve economic or biodiversity-related goals. In particular, much of the recent literature has described large-scale changes to the physical structure and ecology of coastlines associated with urbanisation and coastal development (Airoldi et al. 2005, Firth et al. 2014, Bulleri & Chapman 2015). This can result in loss of habitat (reviewed in Dugan et al.

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2011), loss of native species diversity (Chapman 2003a, Firth et al. 2013, Aguilera et al. 2014), biotic homogenisation through spread of invasive species (Vaselli et al. 2008), changes to relative abundances and ecological interactions among species (Iveša et al. 2010, Klein et al. 2011), reductions in reproductive output (Moreira et al. 2006), loss of habitat for economically important species (Toft et al. 2013), among other changes.

It has been suggested that many of these ecological impacts could be minimised or reversed by using principles of ecological engineering (Mitsch & Jørgensen 2003) when building coastal infrastructure, with ecological engineering defined as 'the design of sustainable ecosystems that integrate human society with its natural environment for the benefit of both' (Mitsch 2012, p. 6). Most projects on ecological engineering of coastal infrastructure have focused on small-scale engineering of novel microhabitats in defence structures, such as seawalls (Chapman & Underwood 2011, Chapman et al. 2017). To date, the microhabitats included in such projects have mimicked relatively simple rock-pools (Browne & Chapman 2014, Firth et al. 2014, Evans et al. 2016), or added small holes and crevices (Firth et al. 2014, Coombes et al. 2015), although large lengths of the Seattle Harbour walls have been extensively modified to create food and shelter for juvenile salmon (Toft et al. 2013). More complex habitat types are currently being considered (e.g. prototype 'Bioblocks'; Firth et al. 2014).

Construction of new areas of habitat can potentially offset degradation to natural communities elsewhere (Hueckel et al. 1989), or target certain species for population enhancement in fisheries (Butler & Herrnkind 1997, Briones-Fourzán & Lozano-Álvarez 2001, James et al. 2007). It must be noted, however, that created habitat seldom entirely mimics natural habitat (Race & Christie 1982) and may take decades to develop full ecological functions (Frenkel & Morlan 1991, Detenbeck et al. 1992, Pratt 1994). With respect to artificial reefs that are constructed to enhance fisheries, there is also concern that their main role is to attract fish away from more natural habitats, concentrating them into habitat of relatively poor quality (Bohnsack et al. 1997). In addition, there is considerable evidence that artificial habitat does not attract and support similar assemblages as natural habitat (Seaman 2007, Macreadie et al. 2011). Thus, the concept of creating natural habitat in order to later be able to degrade or destroy natural habitat is not a suggestion that most ecologists would support.

For a range of benthic invertebrate fisheries, new habitat is constructed for 'sea ranching' which can be defined as 'releasing juvenile specimens of species of fishery importance raised or reared in hatcheries and nurseries into the sea for subsequent harvest at the adult stage or manipulating fishery habitat to improve growth of the wild stocks' (Mustafa 2003, p. 142). To do this, habitat which is used naturally by the target species is constructed, with no consideration normally given to which other species may use it (Bartley & Bell 2008). Target taxa include abalone, sea cucumbers, lobsters, and fish (Bartley & Bell 2008). Some relatively complex 'ranching' habitats have been constructed (e.g. elaborately featured concrete described by McCormick et al. 1994, James et al. 2007) although this is generally done with little understanding of exactly which fundamental environmental variables are important to the targeted species and what the required thresholds of these variables are, before artificial habitat becomes useful.

Theory from the rich history of studies on the factors that promote species diversity in natural benthic habitats (Menge & Sutherland 1976, Sousa 1979, Underwood 2000) can be drawn upon to guide ecological engineering and habitat construction. For example, theory suggests that large benthic species diversity is promoted by increased structural complexity of habitat (McGuinness & Underwood 1986, Archambault & Bourget 1996) and moderate physical instability of substrata (Sousa 1979, McGuinness 1987b). Marine habitats where these 2 features are brought together, and which are particularly relevant to many coastal developments (Green et al. 2012) and fisheries (Shepherd & Turner 1985, Džeroski & Drumm 2003), are intertidal boulder fields and subtidal boulder reefs. Some boulder fields are relatively physically stable over time due to the boulders being closely fitted (Bishop & Hughes 1989) or being too large for waves to move (Nott 2003), but other coastal boulders are periodically moved (McGuinness 1987a). The ecological complexity and instability of these boulder fields has led to them being featured in research that helped frame general theory on ecological succession (Sousa 1979, reviewed by Chapman 2017). Insights about succession are still being drawn from these habitats (Liversage et al. 2014), as well as insights about species diversity (Chapman 2002b, Le Hir & Hily 2005), species invasion (Green & Crowe 2013, Kotta et al. 2016), disturbances and species area relationships (McGuinness 1987a,b), and habitat construction/restoration (Chapman 2012, 2013, Green et al. 2012).

Boulders are defined as unconsolidated rocks >256 mm in maximum diameter (Wentworth 1922; although in this review large cobbles are also generally considered as boulders). Because boulders are easy to quarry, move, and manipulate, they are often used to build coastal infrastructure (e.g. seawalls, riprap, groynes, and gabion baskets), in addition to artificial reefs (see sections below). Boulders are, however, unique intertidal habitats, not replicated by rocky shores (e.g. Wallenstein & Neto 2006); they have potential to therefore provide important intertidal habitat, but little research has been done on how best to construct infrastructure using boulders to create habitat for natural fauna and flora. Ecological engineering may be useful for achieving this goal, especially in semi-sheltered locations, where boulder fields can support a range of rare specialist species (Kangas & Shepherd 1984, Chapman 2012), and great species diversity (Chapman 2002a). Intertidal boulder fields will become more important as habitat when sea levels rise and storms increase, both of which will reduce the area of intertidal habitat, including rocky intertidal habitats such as boulder fields (Jackson & McIlvenny 2011). This will be especially severe where there is no room for habitats to migrate inshore (coastal squeeze; Pontee 2013). The high sensitivity of intertidal boulder habitats to wave energy (Sousa 1979, McGuinness 1987a) means that among rocky habitats, relatively large impacts can be expected to affect boulder fields, and activities such as habitat restoration may become increasingly required.

The research that has already been done on restoration and creation of boulder habitat has demonstrated that in estuaries and on the open coast, the engineered habitat is colonised quickly by a diverse suite of species (Chapman 2012, Støttrup et al. 2017). In addition, the methods required are relatively low-cost and have a high chance of being successful. While there is an intrinsic problem with incorporating a dynamic structure (e.g. a boulder field) into engineered coastlines (where the need is often to create long-lasting and stable shorelines), there are still many potential advantages of using boulders to enhance habitat around engineered structures. Natural habitats associated with armouring (e.g. surrounding soft-sediment or rocky shore) are often heavily impacted by the built structures (Airoldi et al. 2005, Walker et al. 2008); although engineered boulder habitat may replace areas of those natural habitats, the ecological benefits of the created habitat may outweigh the impact caused by habitat replacement (e.g. shifting from impacted soft sediment to biologically diverse boulder field). If impacts from the armouring on natural surrounding habitats are minimal, however, habitat replacement with boulders will likely be inadvisable.

The aim of this paper is to stimulate research and application of knowledge in using boulders for marine ecological engineering and habitat restoration. We reviewed the current understanding of engineered marine boulder habitat, focusing on the lessons learnt from projects of restoration of degraded habitat, deployment of new habitat, and engineered habitat for coastal defences. Future directions are discussed concerning (1) incorporating the factors that support high species diversity in natural boulder reefs into mainstream ecological engineering projects, (2) fine-tuning the artificial provision of habitat for important target species that use boulder reefs, and (3) maximising the benefits from previously demonstrated boulder-habitat construction/restoration.

ECOLOGICAL RESTORATION OF BOULDER HABITAT

Intertidal habitat restoration

Areas of intertidal boulder habitat can be degraded via anthropogenic disturbances such as boulder extraction (Dahl et al. 2009), pollution (Irvine et al. 2006), sedimentation (McGuinness 1987b, Fabricius & Wolanski 2000), and bait collection (Cryer et al. 1987). Also, seawalls and other coastal defences in urban areas have replaced natural shoreline habitats, including intertidal boulder fields (Chapman 2012). The associated loss of species diversity (Chapman 2003a) and of sought-after ecological functions (e.g. fisheries production; Toft et al. 2013) can be reversed by restoring the boulder habitats.

This can be done in ways that are largely passive; for example, boulder habitat can be introduced to the base of seawalls that would otherwise have low habitat diversity (Fig. 1A) by leaving building rubble after construction (Fig. 1B). This would replace some areas of habitat naturally occurring at the base of the seawall (e.g. soft sediment) and effectively produce a third type of habitat (including the seawall). The replacement will only occur, however, along a narrow band metres from the seawall, and may protect the seawall from wave action to some extent (Griggs et al. 1995). Also, construction of boulder habitat near seawalls should only be considered if a high habitat value of the constructed habitat can be demonstra-

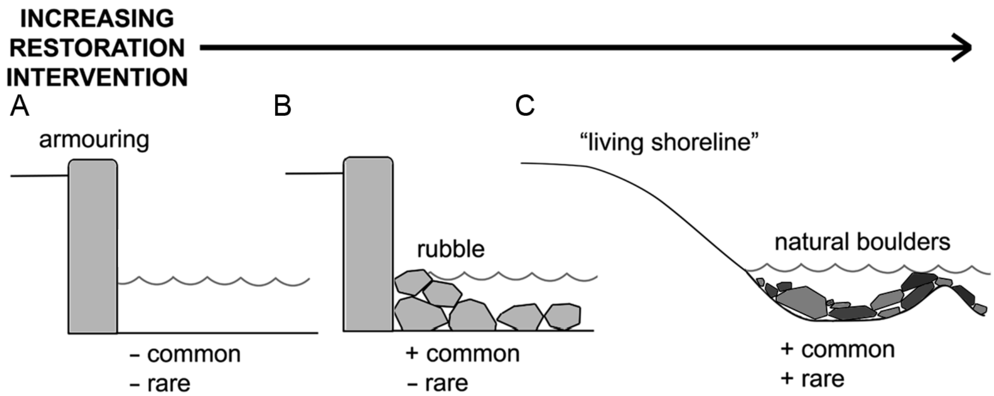

Fig. 1. Levels of intervention for restoration of boulder habitat on shores modified by armouring, and the expected presence/absence of populations of common and rare boulder species in each scenario: (A) no restoration and lack of boulder habitat, (B) passive intervention with placement of building rubble at the base of the construction, and (C) replacement of the construction with a 'living shoreline' (e.g. sand dune, beach, saltmarsh or mangrove; Chapman & Underwood 2011) and restoration of a natural boulder field. This may involve boulders of a different shape to fragments of building rubble and of the naturally occurring rock type(s). A 'bench' may also be included at a distance from the base for increasing habitat use by fish (Toft et al. 2013). In (A) some common boulder species may survive on the armouring but populations can be expected to be limited. Likewise, some rare boulder species may survive in rubble (B) but this has not yet been found (e.g. Chapman 2006). Restored natural boulders (C) consistently harbour extensive populations of common and rare boulder species (Chapman 2012, 2013)

ted, especially if constructed from discarded building rubble. The assemblages developing in this kind of artificial intertidal habitat can be similar to those in naturally occurring boulder fields, although rarer species can be absent (Chapman 2006). As yet unknown features of natural boulder fields may be absent from beds of rubble adjoining seawalls, and these features may be required by some specialist species, which are only known to colonise restored habitat in non-armoured shores. For example, the gaps/interstices underneath and among boulders (Liversage et al. 2017) may be different in constructed and natural boulder fields, and be a feature to which the specialist species respond. Reversion to a 'living' shoreline is the most complete restoration option (Fig. 1C). Examples of restored living shorelines include sand dunes, beaches, mangroves and saltmarshes (Bilkovic et al. 2016). Boulder habitat included in living shorelines is expected to allow inclusion of rare boulder species (e.g. Chapman 2012). To include fish in such restorations, habitat 'benches' can be incorporated (Fig. 1C), which is a method that has been used in marine (Toft et al. 2013) and freshwater (Pander & Geist 2010) restoration projects.

Boulder habitats can also be actively restored by the addition of new boulders to replace those previously extracted from the area, which has been done extensively in some regions to gain material for constructing harbours and coastal defences (Schwarzer et al. 2014, Støttrup et al. 2017). The effectiveness of deploying new boulders for intertidal habitat restoration has been demonstrated, particularly for enhancing species diversity of invertebrates. For example, Chapman (2002a) found that 117 invertebrate taxa colonised new boulders added to existing boulder fields after only 38 d. Additions of new boulders to existing boulder fields have been done to test hypotheses for basic research, but can also provide important information for the practice of restoration by increasing the density of boulder habitat in areas that have been degraded.

Colonisation of new boulders by mobile species generally occurs quickly, but with great variability (Chapman 2002a, 2003b, 2007), mirroring the distributional variability in the natural assemblages (Chapman 2002b, Liversage & Benkendorff 2013). Colonisation can be affected by the size of the newly added boulders (Chapman 2007), the habitat on which the boulders are placed (e.g. sand or algae; Chapman 2002a), the rock type of the boulders (Liversage et al. 2014), and the state of the sessile assemblage already on boulders (Chapman 2003b), but effects are very context dependent (e.g. vary spatially or among different taxa). Many mobile invertebrates migrate frequently across the inter-boulder matrix (Liversage et al. 2012, Liversage & Benkendorff 2017), so whether or not a boulder is positioned adjacent to existing boulders is unimportant for their colonisation (Chapman 2003b).

Large amounts of dispersal also enable rapid colonisation of new, large-scale patches of boulders added near existing boulder fields, which can enhance and restore spatial dimensions of the habitat. After only a few weeks, mobile assemblages in new patches of 50 to 100 boulders were similar to those in nearby existing boulder fields (Chapman 2013), although sessile assemblages can remain different for 1 yr or longer after deployment (Chapman 2012). The size of the added habitat patch can be important for colonisers; some are associated with large additions of boulders and others with smaller additions (Chapman 2013).

Subtidal habitat restoration

When boulder habitat is restored in the subtidal zone, the aim is often to enhance fish populations for reasons related to species diversity or fisheries production. Much research has focused on the habitat value for fish of artificial reefs, many of which are constructed with rock boulders or concrete blocks of various sizes. This research was reviewed in detail by Baine (2001). Here, we reviewed recent research that focused strongly on restoration of natural boulder habitat. For example, projects in the Danish Kattegat have recently aimed to produce restored boulder habitat that closely mimics the original natural boulders, which were extensively extracted within the region (Schwarzer et al. 2014). Large boulders were piled between 2.5 and 6 m high to restore the cavernous boulder reef to the height above the seafloor recorded prior to boulder extraction (Stenberg et al. 2015). Restoration caused economically important fish to spend more time within the habitat than outside (Kristensen et al. 2017) and have greater abundances compared to the same area pre-restoration (Støttrup et al. 2014). Enhanced fish populations in these restored boulder reefs also appeared to increase the frequency of visitations by predators such as dolphins (Mikkelsen et al. 2013).

Another example is from restoration of reefs in the Red Sea (Abelson & Shlesinger 2002). Reef fish populations were successfully enhanced in the degraded reefs by deploying limestone boulders. The size of boulders was important; larger boulders promoted greater fish colonisation (Abelson & Shlesinger 2002). Similarly, addition of large boulders was used to restore reef structures damaged by 2 ship-groundings in Florida (Miller & Barimo 2001). In this case, recruitment of corals was monitored; more species and greater abundances recruited to a site restored with natural boulders compared to a site where boulders were artificially stabilised with cement (Miller & Barimo 2001). Støttrup et al. (2017) provided a summary of considerations that are required for projects of subtidal boulder-habitat restoration, which includes the hydrodynamic and sedimentary conditions of the restoration site, and the capacity of the seafloor to carry the boulders.

Future research on restoration of boulder habitat

While much is known about biotic responses to restored intertidal and subtidal boulder habitats, there are important aspects of their restoration that have not yet been sufficiently researched. These relate to the substratum on which boulders are placed and the engineering of reef structure from layering of boulders. Boulders generally rest on sediment mixtures of varied grain size (fine sand to gravel or pebbles) and these affect the abundances and diversity of biota in the sediment (Cruz Motta et al. 2003). There has been limited research on the interactions between this biota and that occupying the boulders themselves (Cruz-Motta 2005), and this is an important field for future research. For some taxa found on natural boulders, underlying fine sediment is associated with greater abundances, and greater rates of immigration onto the boulder from surrounding areas compared to coarse sediment (Liversage et al. 2012). Boulders can also overlay rock platform, and the value of this substratum relative to sediment is currently unknown. Much of the observed variability in colonisation of restored boulder habitat (Chapman 2002a, 2003b, 2007) may reflect the varied underlying substrata. A number of future research directions would help develop a better understanding of these dynamics; e.g. more studies could pair biotic measurements of under-boulder assemblages with abiotic measurements of the substratum on which boulders were lying. More studies could also perform manipulations of under-boulder conditions to determine biotic responses. These conditions include the gaps, or interstices, underneath boulders, measurements of which could benefit from more fine-scale methods. Such research will be essential to inform future boulder-habitat restoration projects.

Studies on high-shore boulder fields have shown the importance of the number of boulder layers. A double layer creates interstices not present in single layers. Double layers increase species diversity and abundances of some crabs, snails, and limpets due to effects from shading (Takada 1999). While previous subtidal boulder restoration projects have used multiple layering (Støttrup et al. 2017), there have been no comparisons to determine the relative effects of varying this characteristic. Effects will likely involve not only shading, but also introduction of a more cavernous structure (Richter et al. 2001, Alexander 2013). The caverns or gaps provide space for the bodies of animals to fit under boulders where they are protected from the high levels of predation experienced if they become exposed (Shepherd & Clarkson 2001). It is important for theory on boulder-habitat restoration for effects of these caverns/interstices to be better understood.

The demonstrated ecological success and cost effectiveness of restoration, particularly for intertidal/shallow subtidal reefs, points towards the next step as being an increase in the scale of restoration efforts. The recent studies can be considered as 'proof of concept', and large-scale application of the proven methods can be undertaken in situations such as restoration of natural shorelines following dismantling of coastal defences (Toft et al. 2014) and in areas where boulders have historically been extracted.

CREATION OF NEW HABITAT

Boulder habitat created for fisheries

Not all benthic areas are able to maintain a long-term stable field of boulders (Støttrup et al. 2017), but it is still feasible in many coastal areas to create new boulder habitat where none existed previously. New areas of boulders or boulder-like habitat are created in the construction of artificial reefs, and also for sea ranching. Creation of artificial reefs for fish has been reviewed in detail by Baine (2001) and others, so here we focus on sea ranching (Bell et al. 2008), which includes many examples where boulder habitat specifically provides a key requirement. Ranching is most commonly done for abalone fisheries, because juvenile *Haliotis* spp. require protection from predation in sheltered habitat, which is generally boulders (Shepherd & Turner 1985, Read et al. 2013). Novel shelters for fisheries enhancement of abalone have been constructed from artificial materials, such as plastic (McCormick et al. 1994), but most often concrete structures are used. These can be similar in structure to naturally occurring boulders (McCormick et al. 1994), or they can be highly artificial in structure (Davis 1995, James et al. 2007). If retention of natural dynamics is of concern, it is likely that the use of natural habitat types would be most ideal for creation of new habitat. Thus, the addition of rock boulders (Dixon et al. 2006, Roberts et al. 2007, Read et al. 2013) or structures of similar design and/or function (McCormick et al. 1994, Davis 1995) could be considered as the preferred option. Examples of these structures that can be sourced without degradation to marine habitats include quarried boulders (Chapman 2002a), rock pavers/blocks (Chapman 2003b), or concrete structures designed to function similarly to rock boulders (Liversage et al. 2017).

Important factors for creation of new abalone habitat include how the boulders are layered. Survivorship can be greater when multiple layers are created (Dixon et al. 2006), although this effect can be variable (Read et al. 2013). The presence of conspecifics in a newly deployed habitat can make a large difference to subsequent recruitment (Davis 1995), so initial seeding may be useful. Disturbance of substrata by water motion (especially during storms) can cause large reductions in the habitat value of created habitat (Roberts et al. 2007), so the exposure level of deployment locations is an important consideration. Overall, there is increasing evidence that the size and stability of boulders not only affects sessile species assemblages (Sousa 1979, McGuinness 1987b), but also mobile species, such as abalone (McClintock et al. 2007, Roberts et al. 2007, Liversage 2015).

Boulder habitat created for conservation

Boulder habitats have been highlighted as requiring important consideration in programmes of coastal conservation (Thompson et al. 2002, Banks & Skilleter 2007, Rush & Solandt 2017). If such programmes aim to maintain or increase overall species diversity, they may benefit from including the creation of new, biologically diverse (e.g. Chapman 2002b, Le Hir & Hily 2005, Liversage & Benkendorff 2013) boulder habitat. Particular species of conservation concern could also be targeted, as a range of endangered or threatened marine and freshwater species rely on habitats provided by boulders and cobbles (Table 1). For example, some rare and threatened intertidal sea stars (*Parvulastra vivipara* and *P. parvivipara*) are completely reliant on small patches of boulder habitat in isolated intertidal rock pools (Dartnall 1969, Roediger & Bolton 2008, Liversage 2015). Their population sizes may be increased by addition of boulders into empty rock pools, or even creation of new rock pools (Underwood & Skilleter 1996, Evans et al. 2016) with added boulders.

Moreover, creation of new boulder habitat should not only be considered for abalone of commercial importance, but also those of conservation importance, including the Endangered white abalone *Haliotis sorenseni* (Rogers-Bennett et al. 2016). This

Table 1. Shallow-water species that use boulder or cobble habitats during all or part of their life cycles, and are included in the IUCN Red List (for species without an IUCN status, the status from regional conservation listings was used instead). The list includes marine abalone, slugs, chitons and sea stars, and freshwater crustaceans and fish. Boulder habitat would be required as a key consideration during creation of new habitat for all the species

Species name	Species type	Region	Conservation status	Life stage using boulder/cobble habitat	Reference(s)
Haliotis cracherodii	Abalone	North American Pacific coast	Critically Endangered	Juvenile	Miner et al. (2006)
Haliotis kamtschatkana	Abalone	North American Pacific coast	Endangered	Juvenile	Rogers-Bennett et al. (2004), Straus & Friedman (2009)
Haliotis sorenseni	Abalone	North American Pacific coast	Endangered	Juvenile	Lafferty et al. (2004), Rogers-Bennett et al. (2004, 2016)
Haliotis fulgens	Abalone	North American Pacific coast	Species of Concern	Juvenile	Carreón-Palau et al. (2003)
Haliotis corrugata	Abalone	North American Pacific coast	Species of Concern	Juvenile	Carreón-Palau et al. (2003), Rogers-Bennett et al. (2004)
Smeagol climoi	Slug	New Zealand	Critically Endangered	Adult, juvenile	Freeman et al. (2013)
Smeagol manneringi	Slug	New Zealand	Critically Endangered	Adult, juvenile	Freeman et al. (2013)
Bassethulia glypta	Chiton	Southeast Australia	Threatened	Adult, juvenile	O'Hara (2002)
Patiriella vivipara	Sea star	Tasmania, Australia	Threatened	Adult, juvenile	Dartnall (1969)
Euastacus armatus	Crayfish	Southeast Australia	Threatened	Adult	Noble & Fulton (2017)
Galaxias auratus	Fish	Tasmania, Australia	Threatened	Adult	Stuart-Smith et al. (2008)
Galaxias fuscus	Fish	Southeast Australia	Critically Endangered	Eggs	Stoessel et al. (2015)
Galaxias olidus	Fish	Southeast Australia	Critically Endangered	Eggs	O'Connor & Koehn (1991)

species, along with 2 other abalone on the North American Pacific coast (Table 1), is at risk of extinction, largely associated with recruitment failure (Stierhoff et al. 2012). It requires the same boulder habitat for recruitment and survival (Lafferty et al. 2004) as other *Haliotis* species (Shepherd & Turner 1985), and this habitat could be created artificially.

Future research on creation of boulder habitat

Both for species used in fisheries and those included in conservation programmes, further research is required on the small-scale requirements of the rock substrata for colonisation and persistence of populations. The rationale for the designs of concrete structures for new abalone habitat is to mimic the natural interstices among boulders that protect juveniles from predation (McCormick et al. 1994). The importance of the specific characteristics of the interstices in the habitat for juvenile abalone will vary depending on the predators present. Molluscivorous fish, large crabs, and sea stars are particularly effective at causing post-settlement abalone mortality (Read et al. 2013). These predators will be excluded from narrow interstices among substrata such as boulders or cobbles; e.g. Aguirre & McNaught (2013) showed that the presence of cobble refuges increases survivorship of juvenile abalone by a factor of 16 when large predatory sea stars are active. Smaller predators can, however, still reduce abalone recruitment in boulder habitat (Read et al. 2013). Design of the interstices in habitat created for abalone fisheries could be done to target specific problem predators for exclusion.

To our knowledge, only one study has manipulated these interstices and determined the effects on some invertebrates with similar ecologies to abalone (Liversage et al. 2017). Macroinvertebrates including chitons, sea cucumbers, and sea urchins had greater abundances in wide compared to narrow interstices underneath natural boulders and artificial boulders designed to control the properties of the interstices (Liversage et al. 2017). Abalone were not present in these assemblages, but juvenile abalone likely use interstices in a similar way as other molluscs living under boulders, such as chitons. If so, this would provide further evidence that the interstices are a habitat feature requiring important consideration when creating or restoring habitat for abalone.

Another future research direction could be an expansion of the use of boulders for sea ranching of other species. For example, sea cucumbers that do not use refuges suffer heavy predation during sea ranching (Purcell & Simutoga 2008). Efforts could be switched to species that utilise boulders as refuges (Džeroski & Drumm 2003), which could be artificially provided. Similarly, future research could investigate artificial provision of habitat for other taxa that are harvested and which use habitats such as boulders and cobbles, including crabs (e.g. Richards 1992) and sea urchins (e.g. Smoothey & Chapman 2007).

ECOLOGICAL ENGINEERING OF BOULDER HABITAT

Engineering 'soft' coastal defences

Shorelines can be protected from wave action and strong currents by using 'soft' engineering approaches that utilise the barriers to water motion or erosion provided by natural habitats. These barriers are considered to produce a 'living shoreline' when formed of large types of vegetation that dominate a shore (Bilkovic et al. 2016). Various forms of vegetation in the intertidal zone have been studied in this context, e.g. mangroves (Ewel et al. 1998) and saltmarshes (Shepard et al. 2011). Habitats in the subtidal zone have likewise been studied, e.g. seagrasses (Fonseca & Cahalan 1992) and large macroalgae (Løvås & Tørum 2001). While there is considerable interest in stabilising shorelines by relying on natural vegetation, its effectiveness is highly variable, both spatially and temporally (Koch et al. 2009). For example, the biomass of wave-attenuating mangrove, saltmarsh, and seagrass plants can vary drastically among seasons and years, providing unreliable shoreline stabilisation (Koch et al. 2009). While some consideration does need to be given to the stability of boulder reefs (Støttrup et al. 2017), this kind of habitat may be engineered to provide shoreline stabilisation which is more reliable than that provided from living structures, but which is still based on a natural habitat type, unlike most sea defences. Biotic and abiotic components of shore protection are not mutually exclusive (Currin et al. 2010, Bilkovic & Mitchell 2013); any level of incorporation of boulders into living shorelines of vegetation can be expected to increase reliability of shoreline stabilisation.

Fields of different sized boulders in shallow water can attenuate waves approaching the shore (Cusson & Bourget 1997, Guichard & Bourget 1998), and when used as armouring directly on the shore (e.g. 'rip-rap'), they provide useful habitat (Seitz et al. 2006) especially if the overall structure is highly porous (Sherrard et al. 2016). Methods involving engineering of boulder habitat could fill the need to reliably stabilise shorelines while maintaining the ecological functions expected from natural coasts (Barbier et al. 2011).

Engineering 'hard' coastal defences

'Hard' coastal defences are highly engineered structures that provide reduced diversity of habitats and species compared to the natural habitats they replace (Chapman 2003a, Bulleri & Chapman 2010, Gittman et al. 2016). So far, the only coastal defence that has been studied in an ecological engineering context, and which includes small boulders, has been gabion baskets (Firth et al. 2014). Firth et al. (2014) found that effects on abundances of epibiota were caused by different sized boulders within the gabion baskets, but the effects were spatially inconsistent. Porous coastal defences, such as those provided by gabion baskets, do provide useful habitat (e.g. Sherrard et al. 2016); it may be beneficial to prioritise this type of defence (in some circumstances) over others that are less porous and provide fewer types of microhabitats. On the open-coast with increased wave energy, however, gabion baskets are not considered a suitable option (Jackson et al. 2006).

One feature of natural rocky shorelines excluded by 'hard' coastal defences is rock pools, which contain standing water during low tide. This feature is required for highly diverse rock-pool inhabiting species (Martins et al. 2007, Firth et al. 2013). Species that use rock pools can only be introduced into vertical seawalls by incorporating ecological principles into construction practices (e.g. shaded rock pools in constructed cavities; Chapman & Blockley 2009). Alternatively, existing coastal defence structures can be retrofitted by attaching unshaded water-retaining structures (Browne & Chapman 2014), drilling cores into the structures (Evans et al. 2016), adding precast concrete rock pools (Perkol-Finkel & Sella 2015), or constructing concrete rock pools directly on the defences (Firth et al. 2016). The goal is to enhance biological diversity on the artificial structures, which is achieved by increasing species' vertical distributions, i.e. allowing low-shore species to colonise engineered upper-shore areas (Chapman & Blockley 2009). This may be extremely important in areas where, prior to modification, the rocky shore was gently sloping and where the tidal range is not very large. In this situation, the reduction of the area of intertidal habitat caused by modification to a vertically orientated shore (e.g. seawall) may be more than 90%. The relationship between species diversity and area is well documented (McGuinness 1984), so a large reduction in intertidal area by converting gently sloping to steeply sloping habitat will likely increase pressure on intertidal species.

If specific habitat types are targeted, coastal ecological engineering allows not only an increase in species' vertical distributions, but also their horizontal distributions (i.e. along coastlines) on modified shores. Intertidal boulder habitat generally occurs in discrete areas, such as fields of boulders. Horizontal distributions would be increased if boulder habitat was introduced to built shores, which could allow large-scale (i.e. 10s of km) increases in habitat for the associated rare and specialist species (Kangas & Shepherd 1984, Chapman 2012, Liversage 2015). Isolated fragments of boulder habitat sometimes occur naturally within rock pools on rocky shores (e.g. Benedetti-Cecchi & Cinelli 1996, Maggi et al. 2012); if basic rock pools can be engineered on built shores, then engineering of habitat that includes rock pools as well as associated boulders may be possible. While dispersal toward and colonisation of the built boulder habitat would be difficult for some direct developing species (e.g. Liversage 2015), colonisation should not be limited for the majority of boulder species that are larval dispersers (Chapman 2017), provided the engineered habitat meets their specific habitat requirements.

Future research on boulder habitat engineering

The field of ecological engineering of 'hard' coastal defences could benefit from expanded study in a range of research directions related to incorporation of boulder habitat into engineered solutions. For example, while gabion baskets contain a highly modified form of habitat, further research could determine how to better mimic natural boulder habitats in these structures. Other aspects of the boulders used in gabion baskets could also be studied, such as the ecological effects of the rock type used (Liversage et al. 2014) and boulder shape (Liversage 2016).

Regarding seawalls, further research on the habitat provided by boulders or rubble deployed at their bases (Fig. 1b) could be useful, including how these structures increase habitat complexity and potentially dampen the effects of wave action. Before habitat that effectively mimics natural boulders can be effectively included in 'hard' coastal defences, however, research is needed on the habitat requirements of the boulder-specialist species. Their requirements can be relatively subtle (e.g. Liversage et al. 2017), and much of the variability that occurs in the distribution of species among individual boulders and boulder fields has not yet been explained (Grayson & Chapman 2004, Chapman 2017). But as our understanding of the ecology of boulder habitat increases it will become more feasible to develop urban shorelines that include the highly novel elements of biological diversity represented by the rare species that live in association with boulders.

CONCLUSIONS

As the impacts increase from rising sea levels (Thompson et al. 2002) and amplified wave energy (Reguero et al. 2015), the pressure on coastal land managers to modify shorelines will increase. Effective coastal management requires that the modifications minimise impacts to coastal species diversity and the functions provided by intertidal and subtidal communities. This review has shown that there are many opportunities available for using boulder habitat as a tool in the field of coastal ecological engineering. In some regions where boulder habitats are scarce, they could be engineered to increase amounts of this rare habitat type and associated specialist species, while on coasts where boulder habitats are common, including boulders in engineered shorelines may still be beneficial. In addition, this review has highlighted the benefits of in-depth consideration of habitat requirements for managing species of fisheries or conservation value that use boulder habitat.

The species that use boulder habitat are largely hidden (i.e. located underneath the boulders or in narrow interstices). Thus, they are not widely considered by coastal researchers, nor by managers of coastal habitats. But as knowledge about this group of species increases, the applications for that knowledge, such as those described in this review, will enable a unique and valuable element of biological diversity to be incorporated into programmes of coastal management and ecological engineering.

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