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DESIGN OPTIONS, IMPLEMENTATION ISSUES AND EVALUATING SUCCESS OF ECOLOGICALLY ENGINEERED SHORELINES

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Abstract

Human population growth and accelerating coastal development have been the drivers for unprecedented construction of artificial structures along shorelines globally. Construction has been recently amplified by societal responses to reduce flood and erosion risks from rising sea levels and more extreme storms resulting from climate change. Such structures, leading to highly modified shorelines, deliver societal benefits, but they also create significant socioeconomic and environmental challenges. The planning, design and deployment of these coastal structures should aim to provide multiple goals through the application of ecoengineering to shoreline development. Such developments should be designed and built with the overarching objective of reducing negative impacts on nature, using hard, soft and hybrid ecological engineering approaches. The design of ecologically sensitive shorelines should be context-dependent and combine engineering, environmental and socioeconomic considerations. The costs and benefits of ecoengineered shoreline design options should be considered across all three of these disciplinary domains when setting objectives, informing plans for their subsequent maintenance and management and ultimately monitoring and evaluating their success. To date, successful ecoengineered shoreline projects have engaged with multiple stakeholders (e.g. architects, engineers, ecologists, coastal/port managers and the general public) during their conception and construction, but few have evaluated engineering, ecological and socioeconomic outcomes in a comprehensive manner. Increasing global awareness of climate change impacts (increased frequency or magnitude of extreme weather events and sea level rise), coupled with future predictions for coastal development (due to population growth leading to urban development and renewal, land reclamation and establishment of renewable energy infrastructure in the sea) will increase the demand for adaptive techniques to protect coastlines. In this review, we present an overview of current ecoengineered shoreline design options, the drivers and constraints that influence implementation and factors to consider when evaluating the success of such ecologically engineered shorelines.

Introduction and the history of ecological engineering of shorelines

Humans have been altering coastlines for millennia (Thompson et al. 2002, Dugan et al. 2011, Ellis 2015, Loke et al. 2019), usually with little regard for the environment. Natural habitats have been replaced with artificial structures to create access, reclaim land for agriculture, industry, transport, residential use and tourism (Gittman et al. 2015, Lai et al. 2015, Chee et al. 2017) and protect growing coastal populations (Crossett et al. 2004, Nicholls et al. 2007, Firth et al. 2016b) from erosion and flooding (Burcharth et al. 2007, Kittinger & Ayers 2010, Hinkel et al. 2014). Impacts on coastal ecosystems from such developments are inevitable and range from chronic to catastrophic

(Airoldi et al. 2005a, Martin et al. 2005, Bulleri & Chapman 2010, Dethier et al. 2016, Bishop et al. 2017, Heery et al. 2017). Improving environmental conditions along artificial shorelines is critical to compensate for impacts from ocean sprawl—the proliferation of artificial structures in the marine environment (Dafforn et al. 2015a, Dyson & Yocom 2015, Firth et al. 2016b, Bishop et al. 2017, Heery et al. 2017). With climate change and sea level rise, existing shoreline protections are increasingly proving ineffective and will need substantive reassessment and reconstruction during the coming decades (Hawkins & Cashmore 1993, Nicholls & Tol 2006, Hallegatte et al. 2013, Hinkel et al. 2014, Hoggart et al. 2014, Smith et al. 2017a).

The goal of this review is to provide a framework for selecting, applying and tracking the success of ecologically engineered (hereafter 'ecoengineered') shoreline strategies (Mitsch 2012) for intertidal and shallow subtidal marine environments. The fundamental aim of ecoengineered shorelines is to build more inclusive, resilient and safe coasts for people and nature, which maximise benefits for ecosystems, society and economies (Airoldi et al. 2005a, Dafforn et al. 2015a). Because it is a relatively young discipline and innately experimental in its current form, ensuring that it is developed and applied responsibly requires that parameters and metrics for its success be clearly defined and monitored (Saleh & Weinstein 2016, Mayer-Pinto et al. 2017). In this review, we take a multidisciplinary approach in evaluating the benefits and challenges of ecoengineered shorelines and discuss metrics to evaluate success originating in the fields of coastal ecology, engineering, sociology, economics, urban planning and architecture.

Ecoengineered shorelines are in a sense the most recent stage in the development of marine infrastructure and coastal armouring, which can be traced back to before the Common Era (i.e. BC) in Egypt (Loke et al. 2019). Early ports were located in sheltered bays, river mouths and lagoons, with simple jetties and, eventually, low-crested breakwaters used in combination with natural habitats for coastal protection (Polanyi 1963, Hoyle 1989, Charlier et al. 2005). In some regions, such as China and parts of the Mediterranean, technological advances led to more extensive artificial shorelines relatively early (Franco 1996). For instance, with the discovery of pozzolanic ash hydraulic cement, the Romans started extensive underwater engineering and managed to construct solid breakwaters to protect fully exposed harbours (Jackson et al. 2017). By the Middle Ages in Europe, planting and dune belt protection were well established, as were strict environmental regulations. For instance, legal documents of 1282 and 1339 in Venice prohibited the cutting of coastal trees, picking mussels and removing sand or vegetation from beaches or dunes (Grillo 1989). In the Renaissance, Leonardo da Vinci championed the credo of 'working with Nature' ('*ne coneris contra ictum fluctus: fluctus obsequio blondiuntur*'—Nature should not be faced bluntly and challenged, but wisely circumvented) (Franco 1996). A similar idea was championed by Andries Vierlingh in his manuscript entitled *Tractaet Dyckagie*, which remains an important text regarding fundamental errors in land and water engineering management (Vierlingh et al. 1920).

The formal framing of the concept of ecoengineering is generally attributed to H. T. Odum, who proposed that ecosystems provide the biological 'design' and 'energy' required to 'engineer' socioeconomic benefits of human development (Odum 1975). He stated that 'the management of nature is ecological engineering, an endeavour with singular aspects supplementary to those of traditional engineering' (Odum 1971). Since then, the concept has gradually expanded, replacing purely 'ecological processes' with 'natural processes' to now include physical and other factors such as 'design' and 'energy' inputs (e.g. the 'sand engine/sand motor' approach for beach nourishment and flood protection that is used along the Dutch coastline; Mulder & Tonnon 2011, Stive et al. 2013, de Schipper et al. 2016). During the last 10–15 years, the practitioners involved in designing coastal defences have embraced a 'design with nature' approach, which has led to various philosophies and programmes for the delivery of ecologically minded coastal and maritime infrastructure (e.g. the programme of 'working with nature' organised by the World Association for Waterborne Transport Infrastructure (PIANC): https://www.pianc.org/working-with-nature US Army Corps of Engineers' 'engineering with nature': https://ewn.el.erdc.dren.mil/ EcoShape's 'building with nature': www.ecoshape.org).

This transition in perception and practice has been facilitated by the widening recognition of the pervasive and persistent anthropogenic changes that characterise urban areas (Bulleri & Chapman 2010, Firth et al. 2016b, Mayer-Pinto et al. 2018, Bugnot et al. 2019) and by the emergence of the 'novel ecosystem' concept, which emphasises novelty and human agency in the emergent ecological communities that are without historical precedent (Milton 2003, Hobbs et al. 2006). The novel ecosystem concept is not without controversy (e.g. Woodworth 2013, Murcia et al. 2014); however, in novel urban habitats specifically, it provides a framework for conceptualising coastal engineering projects as opportunities for the amelioration of ecosystems that are already heavily shaped by anthropogenic activity (Perkins et al. 2015, Aguirre et al. 2016). While the distinction is not always made, urban areas contain both hybrid ecosystems that may have significant novelty, as well as novel ecosystems, which, by definition, cannot return to hybrid or historical conditions (Hobbs et al. 2006). The novel ecosystem concept has arguably changed the perceived value of the urban environment, particularly in relation to its provisioning of ecosystem services to urban populations, ultimately facilitating the integration of a broader range of ecological, engineering, social and economic interests into shoreline development and management (Perring et al. 2013).

There is considerable overlap between the fields of ecoengineering and ecological restoration. Ecological restoration aims to assist the recovery of degraded ecosystems and put back their attendant services (Society for Ecological Restoration International Policy Position Statement (SER 2004). The traditional view of restoration envisaged returning an ecosystem to a predefined historical state (Hobbs & Norton 1996). Given ecosystem variability and complexity, coupled with the extent of environmental change that has occurred over recent decades, making target baselines uncertain (Palmer et al. 2016), restoring ecosystems to their former state is often impossible on human timescales (Hobbs et al. 2006). Thus, many have emphasised the importance of defining restoration to a defined target state, which is likely to be far from the original pristine condition of the ecosystem (Hawkins et al. 1999, 2002, SER 2004, Geist & Hawkins 2016, Palmer et al. 2016). Even our best efforts serve primarily to rehabilitate or repair damaged biodiversity and ecosystem processes and services, rather than comprehensively restore ecosystems (e.g. in a marine context, 'nudging nature'—Hawkins et al. 1999, 2002). Conversely *reallocation* is the process of assigning an ecosystem to a new use that may not necessarily bear an intrinsic relationship to the structure or functioning of the predisturbed ecosystem (Aronson et al. 1993). *Reconciliation* is the process of modifying and diversifying anthropogenic habitats to harbour a wide variety of native species (Rosenzweig 2003a; for further discussion, see the section entitled 'Links to theoretical and community ecology', later in this review). While some refer to restoration as the best form of ecoengineering (Bradshaw 1997), others claim that restoration encompasses ecoengineering (SER 2004). Here, we propose that ecoengineering is a broad approach that can aid restoration (Mitsch 2012), but more typically is used for rehabilitation, reallocation and/or reconciliation.

During the last decade, ecoengineered shorelines have emerged as a promising alternative to traditional coastal development (Mitsch 2012), although they are by no means a panacea for countering the negative impacts of shoreline construction, and numerous uncertainties regarding their efficacy remain (Bouma et al. 2014, Sutton-Grier et al. 2015). We emphasise that these are not a substitute for natural or even seminatural systems. Neither should such ecoengineering approaches be used to legitimise developments in sensitive habitats. In this review, our focus is on coastlines that have already become urbanised or heavily modified by land reclamation for agriculture, industry or transport infrastructure, including low-density, residential suburban sprawl. The approaches suggested here are particularly suitable when current coastal developments are being expanded or repaired. While there are examples of ecoengineered shorelines around the globe and many under development (Chapman & Underwood 2011, Firth et al. 2014, Elliott et al. 2016, Munsch et al. 2017a), a cohesive framework for developing ecoengineered shorelines and evaluating their success is presently lacking (but see relevant work by van Slobbe et al. 2013, van der Nat et al. 2016, Osorio-Cano et al. 2017, Gracia et al. 2018, Whelchel et al. 2018). This review presents the underpinning concepts, key approaches, recent developments, limitations and future trajectory of ecoengineered shoreline design. We present a framework for the development of ecoengineered shorelines by reviewing three themes: (1) core principles and approaches to ecoengineered shoreline design, (2) factors that influence implementation and (3) how to evaluate the success of ecoengineered shorelines.

Design considerations for ecoengineered shorelines

Links to theoretical and community ecology

Ecological concepts and theory provide both a vocabulary and a predictive framework to understand and manage ecosystems along the spectrum from near-pristine to highly degraded ecosystems. Thus, theory can inform policy, planning and practice. Here, we briefly introduce ecological knowledge relevant to the design of ecoengineered shorelines. We start with species-area relationships and the relationship between biodiversity and ecosystem functioning. We then discuss diversity at different scales, succession and disturbance, environmental gradients and connectivity. The aim of this section is simply to highlight some of the key theoretical underpinnings of the rapidly emerging field of designing ecologically sensitive shorelines.

Species-area relationships

Species-area relationships (SARs)—which involve one of the few widely accepted laws in ecology (Lawton 1999)—dictate that a larger area will contain a larger number of species. This pattern is observed at almost every spatial scale (Figure 1; Rosindell & Cornell 2007, O'Dwyer & Green 2010). It is manifest in highly modified shorelines, where the loss of natural habitat leads to direct loss in species (Rosenzweig 2003a). The same principle may be applied in reverse, however, to aid modifying, redesigning and diversifying anthropogenic habitats. Reconciliation ecology, which focusses on how urban and novel habitats may be used to maintain biodiversity and to provide ecosystem services, provides several examples of this (Hobbs et al. 2013, Perring et al. 2013), in both terrestrial (Rosenzweig 2003b, Kowarik 2011) and aquatic (Mitsch & Jørgensen 2004, Firth et al. 2016b) environments. While species colonisation in some of these examples occurs by accident (refer to Rosenzweig 2003b), many other examples are carefully planned and managed (see the section entitled 'Case studies and scaling up', later in this review).

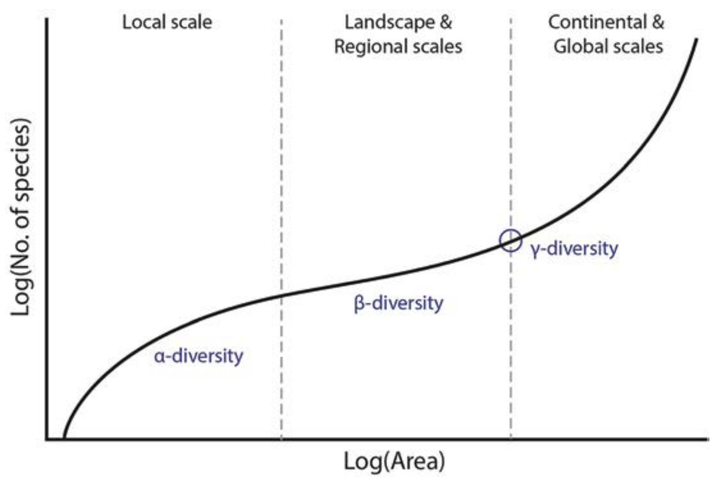

Figure 1 Triphasic SAR. A larger area will include a larger number of species. On a local scale, alpha (α) diversity describes the number of species within a habitat, while the compositional difference in species assemblages among habitats (beta, β, diversity) is essential for achieving high gamma (γ) diversity in a region.

Diversity at multiple scales

Diversity is a key theme in ecology, which underpins both the goals and assessment of many ecoengineered shoreline projects. It is frequently estimated as species richness (i.e. the total number of species present) and can be quantified at multiple spatial scales. While species richness within a habitat (alpha diversity) is useful in many instances, the compositional difference in species assemblages among habitats (beta diversity) is essential for achieving high diversity in a region (gamma diversity; Figure 1; Whittaker 1960, 1972). Specifying the scale at which ecoengineered shoreline designs aim to enhance diversity is crucial, as strategies for improving alpha and beta diversity may differ. Designs focussed on enhancing large-scale (landscape and regional scale—hundreds of metres—tens of kilometres) spatial heterogeneity and creation of various medium-scale habitat types (1–100 m) will have a positive effect on beta diversity. Those strategies focussed on increasing local-scale (<1 m) topographic and/or structural complexity (e.g. Coombes et al. 2015, Loke and Todd 2016) and ameliorating abiotic stressors (e.g. Browne & Chapman 2011, Evans et al. 2016) are more likely to have a positive effect on alpha diversity. Patches of more complex habitat, especially if they are of different types, among a background landscape of less complex habitats will also increase beta diversity (Firth et al. 2014, Evans et al. 2016).

Biodiversity and ecosystem functioning

Over the last two decades, there has been a growing realisation that loss of species can compromise ecosystem functioning (e.g. biomass production, nutrient uptake; Loreau et al. 2002, Tilman et al. 2014, Oliver et al. 2015) and in turn, ecosystem services (e.g. carbon storage; Díaz et al. 2006, Cardinale et al. 2012, Isbell et al. 2014). Thus, increasing diversity would enhance the functioning of ecosystems by elevating production and nutrient cycling, as well as conveying resilience to environmental change (the insurance hypothesis; Yachi and Loreau 1999) and invasion by nonnative species (Stachowicz et al. 2002, Arenas et al. 2006, Tan et al. 2018). These processes lead to enhanced goods and services provided by biodiversity and ecosystems (i.e. provisioning, supporting, and regulating; Raffaelli et al. 2002). For both sedimentary (e.g. Solan et al. 2004) and rocky shore habitats (e.g. O'Connor & Crowe 2005, Griffin et al. 2008, 2010), experimental work has shown that species diversity and identity influence ecosystem processes such as productivity and nutrient cycling. The importance of habitat patch diversity has also been recognised for some time (e.g. Giller et al. 2004, Hawkins 2004) and more recently has been experimentally demonstrated (e.g. Griffin et al. 2009, Godbold et al. 2011, Alsterberg et al. 2017). Knowledge of the traits of species (functional diversity) can also be valuable in translating structural attributes of biodiversity into the maintenance or restoration of ecosystem processes, and hence services (Rigolet et al. 2014). Ultimately, ecoengineering aims to maximise the ecosystem services that can be delivered in seminatural and highly modified ecosystems (Dafforn et al. 2015a). Paying attention to increasing habitat patch diversity, and hence beta species diversity, is likely to pay greater dividends in terms of ecosystem functioning and provision of services.

Environmental gradients and connectivity

Environmental gradients are a major influence on community composition and structure in shoreline environments (see Raffaelli & Hawkins 1996 for a review). The major gradients can be viewed as being either vertical (e.g. tidal elevation/bathymetry) or horizontal (e.g. exposure to wave action). These gradients ultimately determine what species can potentially survive in an area, with biological interactions usually proximately determining the actual assemblage in a location [e.g. experimental work by Jonsson et al. (2006) on the respective roles of wave action and grazing on breakwaters]. Disturbance regimes also vary along these gradients, but the strength and direction of these effects are frequently modified by local contingency (Underwood & Chapman 1996, Chapman et al. 2010, Bulleri et al. 2011), including small-scale topography (Johnson et al. 1998, 2003, Berntsson et al. 2000) and the vagaries of settlement by algal propagules and the larvae of invertebrates (Butman 1987, Underwood & Fairweather 1989, Fletcher & Callow 1992, Knights et al. 2012). Designs that minimise the extent to which environmental gradients are altered or intercepted are less likely to fundamentally change community structure. For example, installing hard coastal armour in an area naturally typified by coarse sand or gravel will inevitably lead to scouring of any colonising rocky shore species (Moschella et al. 2005), preventing communities from developing beyond the early successional stages.

Artificial structures often support assemblages analogous to rocky reefs jutting out from high-energy sandy beaches (Bally et al. 1984) that tend to be dominated by ephemeral early successional species. Additionally, designs perform best when they account for ecological connectivity at various scales. Changes to the movement of organisms and resources such as detritus should be avoided (Huijbers et al. 2013, Bishop et al. 2017). Arrays of structures scaling up to affect whole coastlines (e.g. Ma et al. 2014, Dong et al. 2016) can act as stepping stones for the spread of native species responding to climate change (Johannesson & Warmoes 1990, Mieszkowska et al. 2006, Keith et al. 2010, Firth et al. 2015, Huang et al. 2015, Dong et al. 2016), as well as help invasions by nonnative species (Floerl et al. 2009, Mineur et al. 2012, Airoldi et al. 2015, Bishop et al. 2017).

Succession and disturbance

Novel habitats are colonised by a variety of marine organisms (Wahl 1989). When relatively constant conditions prevail, communities in novel habitats gradually shift in composition and structure through the processes of immigration and extinction (MacArthur & Wilson 1967). Connell & Slatyer (1977) proposed three models of succession: the classical model of positive facilitation of later stages by earlier colonisers, inhibition of later stages by earlier arrivals and a neutral tolerance model reflecting arrival of propagules and longevity of species. Significant disturbances, whether natural or anthropogenic, tend to push a successional sequence back to earlier stages (i.e. system retrogression; Odum 1985). Biological interactions such as grazing have been shown to break inhibition by intermediate phases such as green algae (Sousa 1979).

Classical ecological theory predicts the highest levels of diversity under moderate levels of disturbance, either in terms of intensity or frequency [the intermediate disturbance hypothesis (IDH); Connell 1978], as a low-disturbance regime leads to dominance by competitive-dominant species and strong frequent disturbances suppress most species to also result in low overall diversity (e.g. Sousa 1979). Although IDH has been challenged on both a theoretical and empirical basis (Lubchenco 1978, Fox 2013), it is well established that high levels of disturbance lead to depauperate systems comprising relatively few early successional or opportunistic taxa (Pulsford et al. 2014). As disturbances are commonplace in human-dominated environments, ecoengineering strategies that help stabilise the disturbance intensity or frequency may be beneficial (Airoldi et al. 2005b, Bulleri & Airoldi 2005, Airoldi & Bulleri 2011, Bracewell et al. 2013, Salomidi et al. 2013), increasing the likelihood that communities progress to a climax (Hawkins et al. 1983) or oscillate to and from intermediate stages.

Additionally, species or groups of species that positively influence succession can be integrated into ecoengineered shorelines through seeding (e.g. Perkol-Finkel et al. 2012, Ferrario et al. 2015, Ng et al. 2015) or targeted designs (Strain et al. 2017a). Studies on the mechanisms of succession have highlighted the importance of certain species interactions in determining the rate and direction of community development (Maggi et al. 2011). Positive interactions such as facilitation [e.g. settlement by mussels or tubicolous polychaetes enhancing the colonisation of meiofauna on hard surfaces (Dubois et al. 2002, O'Connor & Crowe 2007) or increased accumulation of leaf litter by mangrove on previously bare substrata, which aids in recruitment of juvenile nekton] may be particularly important (Bulleri et al. 2018). Organisms that serve as ecosystem engineers and foundation species also have profound impacts on the way that communities develop and, ultimately, on diversity as well (e.g. Coleman & Williams 2002, Tolley & Volety 2005, Marzinelli et al. 2014).

Biological invasions

Finally, theoretical and community ecology concepts from invasion biology are also relevant for ecoengineered shoreline design, as artificial structures appear particularly prone to colonisation by nonnative species (Bulleri & Airoldi 2005, Glasby et al. 2007, Vaselli et al. 2008, Ruiz et al. 2009, Dumont et al. 2011, Dafforn et al. 2012, Mineur et al. 2012, Simkanin et al. 2012, Airoldi et al. 2015). These concepts are strongly interrelated with many of those already presented in this review, particularly diversity, connectivity, succession and disturbance. Artificial structures provide novel colonisation opportunities for nonnative species (Dafforn et al. 2012). Those located in urban areas may be subject to particularly high levels of propagule pressure ('introduction effort'), given their close proximity to known invasion hubs such as marinas and shipping facilities (Seebens et al. 2013), which is thought to strongly influence invasion risk (Lockwood et al. 2005). Urban artificial structures may also be subject to particularly high rates of disturbance (Bulleri & Airoldi 2005, Piola & Johnston 2008, Kenworthy et al. 2018) or to relatively low levels of predation by mobile consumers (Simkanin et al. 2013, Astudillo et al. 2016, 2018, Rogers et al. 2016), ultimately favouring nonnative taxa. Further, additional structural complexity, which is a key component of many ecoengineered shoreline projects (discussed further in section 'Environmentally sensitive hard defences', later in this review), can modify predator-prey interactions and potentially facilitate nonnative prey species by enhancing refugia from native predators (Barrios-O'Neill et al. 2014).

Fundamental design options

The construction of artificial structures such as seawalls, groynes and breakwaters has historically been the default approach to coastal protection (Cooper et al. 2016). Artificial structures are economically and environmentally costly (Jones 1994, Airoldi et al. 2005a, Kittinger & Ayers 2010, Hinkel et al. 2014), however, contributing to the growing interest in alternatives that compensate for the negative impacts of traditional coastal infrastructure and provide multiple functions (Chapman & Underwood 2011, Francis & Lorimer 2011, Mitsch 2012, Evans et al. 2017). These alternatives can be categorised into three fundamental approaches: (1) building in design features in new or modifying existing artificial structures (termed *hard ecological engineering*; Chapman & Underwood 2011, Firth et al. 2014); (2) replacing artificial structures with sediments, vegetation and/or other habitat-forming organisms (called *soft ecological engineering*; Temmerman et al. 2013, Morris et al. 2018b) and (3) applying both soft and hard engineering approaches in combination (known as *hybrid ecological engineering*; Chapman & Underwood 2011, Bilkovic & Mitchell 2013) (Figure 2).

Many terms are used to refer to both traditionally and ecologically engineered shores. Common synonyms for artificial structures and coastal defences include coastal infrastructure, shoreline armouring, hard structures and urban structures. The term *Ecoengineered shorelines* is used to describe shorelines that have been developed or retrofitted using ecoengineering principles, via either hard, soft or hybrid approaches. Created or restored habitats are those that have undergone soft ecoengineering, and are commonly referred to as *nature-based coastal defence* (van der Nat

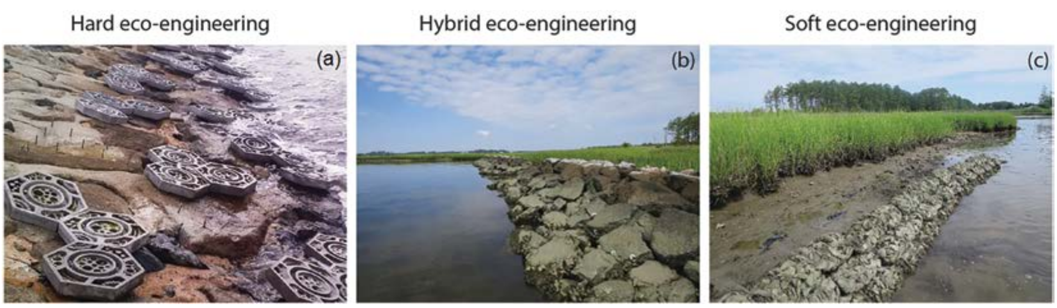

Figure 2 Approaches to ecoengineered shorelines include (a) hard (tile-enhancement units installed along the seawalls at Changi, Singapore), (b) hybrid (rock fillet with saltmarsh in Chesapeake Bay, United States) and (c) soft ecological engineering (bagged oyster shell reef with saltmarsh in Chesapeake Bay).

et al. 2016, Osorio-Cano et al. 2017, Gracia et al. 2018, Sutton-Grier et al. 2018). In the United States, soft and some hybrid ecological engineering projects also fall under the commonly used term *living shorelines* (Bilkovic & Mitchell 2017).

Although the overall objective of ecoengineering is similar regardless of approach (i.e. to build shorelines for the benefit of both humans and nature; Mitsch 2012), there is a fundamental difference between hard and soft ecoengineering from a coastal defence perspective (Morris et al. 2018b). The goal of hard ecoengineering is usually to compensate for the negative impacts of artificial structures (e.g. seawall or breakwater) through enhancing biodiversity and ecological functioning while maintaining their physical integrity (e.g. Moschella et al. 2005, Burcharth et al. 2007, Chapman & Underwood 2011, Firth et al. 2014, Pioch et al. 2018). Conversely, while soft ecoengineering is usually proposed from a conservation and/or sustainability perspective (Dafforn et al. 2015a, Mayer-Pinto et al. 2017), the created or restored habitat needs to provide adequate coastal defence in addition to ecological and landscape values (e.g. biodiversity enhancement) if this technique is to replace or complement artificial structures. Although hard ecoengineering is less challenging to implement from a coastal defence perspective, there is growing evidence that created or restored habitats using soft engineering can also provide protection that is equivalent to or better than traditional engineered hard structures (Gittman et al. 2014, Smith et al. 2017a, Smith et al. 2018). As mentioned previously, soft engineering approaches and hard structures may even operate synergistically to enhance coastal protection (Smallegan et al. 2016, Vuik et al. 2016)—as successful hybrid schemes in some cases.

Goals and objectives

Defining the objectives or goals is considered the most fundamental step in any restoration (e.g. Hobbs & Norton 1996, SER 2004, McDonald et al. 2016, Palmer et al. 2016) or ecoengineering project (Mayer-Pinto et al. 2017). Ecoengineered shorelines are multifunctional and are valued for their potential to accrue benefits across multiple stakeholders (Evans et al. 2017), each of whom has their own set of objectives (e.g. economic, engineering, social, ecological) that typically drive or influence the objectives of a project. Sometimes these objectives overlap—for instance, creating a wide beach could also generate space for recreation, protect inland infrastructure and form habitat for nesting shore birds (Temmerman et al. 2013). Due to the way in which projects are conceived and funded (see the section entitled 'Implementation of ecoengineered shorelines', later in this review), one or more of these objectives may take precedence over the others, which can lead to trade-offs. Nevertheless, multifunctionality is a key goal of ecoengineered shorelines, and thus engineering, ecological and socioeconomic outcomes should be optimised. Evaluation of the success or failure of projects needs to focus on whether these multiple objectives are achieved (see the section entitled 'Evaluating the ecoengineered shorelines approach in practice', later in this review).

Identifying the approach

Ecoengineered shorelines range considerably in form and function and may be conceptualised along a spectrum (Moosavi 2017), from created (i.e. a habitat not historically present) or restored natural habitats at one end to ecologically engineered hard structures at the other. The most suitable approach for a site will depend on socioeconomic variables and the physical, environmental and the ecological contexts (Figure 3; Spalding et al. 2014). Shorelines should be defended only where there is something to protect (Airoldi et al. 2005a). This is usually property or infrastructure, or in some instances, ecological systems (Moody et al. 2016) or archaeological or cultural heritage (Reimann et al. 2018) threatened by inundation or erosion (Figure 3). Land reclaimed for agriculture via empoldering almost always needs some form of protection (Woltjer & Al 2007). Any form of shoreline construction or modification will pose inherent risks to marine communities and the coupled natural-human systems of which they are a part. The need for shoreline protection, therefore, must be sufficiently great to warrant assuming such risks (Airoldi et al. 2005a).

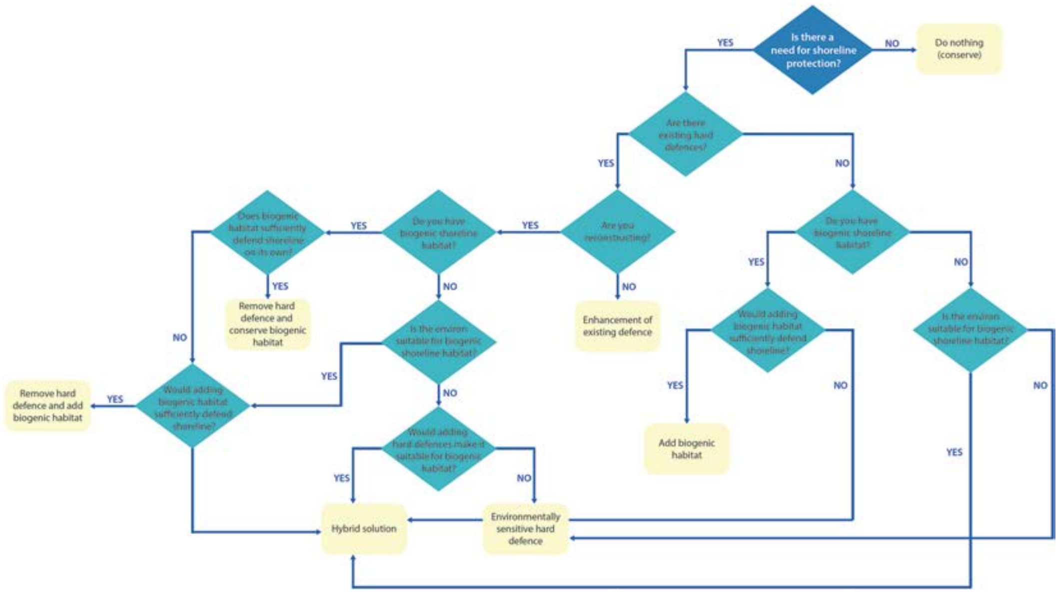

Figure 3 A flowchart for identifying the suitable approach to ecoengineered shoreline protection. Blue diamonds indicate a decision (dark blue is the starting point), and yellow boxes indicate the ecoengineered shoreline approach to take. Biogenic habitat refers to natural or restored (i.e. soft ecoengineering) habitats. *Environmentally sensitive hard defence* refers to hard ecoengineering approaches.

In general, selecting from soft ecoengineering, a hybrid approach and environmentally sensitive hard defences requires consideration of the physical environment (particularly wave conditions; Leonardi et al. 2016) and the respective cost of each approach (Arkema et al. 2017) in the context of the main purpose of the project (e.g. creating a bathing beach or a surfing bay, as opposed to protecting an agricultural land or an ecosystem of conservation interest). At low-wave-energy sites, soft and hybrid ecoengineering approaches are often feasible (Leonardi et al. 2016, 2018, Sanford & Gao 2018). At higher-energy sites, with significant wave action from either wind or shipping, certain types of soft engineering may not persist over time or provide adequate shoreline protection (Bouma et al. 2014). Such cases necessitate hybrid approaches, or at the highest energy end of the spectrum, hard defences (Moosavi 2017).

Despite growing evidence suggesting that soft and hybrid approaches are often cost effective and sufficient for protection from flooding and erosion (Spalding et al. 2014, Gittman & Scyphers 2017, Sutton-Grier et al. 2018), considerable uncertainties (both perceived and realised) remain surrounding their efficacy as coastal defences (Gedan et al. 2011, Perkins et al. 2015, Sutton-Grier et al. 2015, Arkema et al. 2017). Thus, at least some degree of hard ecoengineering is often required (Cooper et al. 2016). The value and/or importance of the property, infrastructure or ecosystem to be protected will have some bearing on the choice of approach (soft, hard or hybrid). For example, if the asset to be protected is essential to national or regional infrastructure, such as an airport, hospital or railway, there may be less willingness to accept (WTA) approaches perceived to carry greater uncertainty with respect to inundation and flood risk. Often, there is stronger support for hard engineering approaches (Gray et al. 2017) that have been more thoroughly tested over time (Sutton-Grier et al. 2015), even when they are unnecessary or less cost effective than soft or hybrid alternatives (Scyphers et al. 2014, Gittman & Scyphers 2017). In contrast, where the asset is less crucial for social and economic health or primarily of local significance, there may be a higher chance for adopting more experimental (and potentially riskier) ecoengineered shoreline approaches that provide protection under most (but not the most severe) storm events, or for which the evidence of efficacy in terms of shoreline protection is limited (but see Gittman et al. 2014, Smith et al. 2018).

Which ecoengineered shoreline approaches can and should be adopted will largely be influenced by the degree to which the environment has already been modified by anthropogenic activities (Bouma et al. 2014). In heavily modified environments, environmental degradation (e.g. by pollution and land reclamation) may result in conditions that are no longer suitable for the establishment of soft or hybrid ecoengineered shorelines (Chee et al. 2017), necessitating use of hard defences, at least in the interim, if cleanup efforts or long-term mitigation of environmental conditions are part of the development plan. The implementation of soft or hybrid approaches may also be restricted by the lack of available space, particularly in heavily urbanised settings (Saleh & Weinstein 2016). In contrast, in environments where intact and functional habitat-forming species are present, the priority should be to protect these and, where necessary, enhance their efficacy in protecting shorelines by increasing their habitat area or encouraging growth forms that are more effective in stabilising sediments, dissipating wave energy or both. Particularly at sites where the natural habitat type is sedimentary (e.g. seagrass beds, mangrove forests, saltmarshes, and dune systems), use of hard defences should be applied only where absolutely necessary, as they result in a qualitative state change from a soft- to hard-bottomed systems (Airoldi et al. 2005a).

Furthermore, the selected approach will also depend on site-specific factors such as tidal range and whether the shoreline is natural or reclaimed land. These factors will influence the space available for interventions, as well as the size and configuration required for an intervention to protect from erosion and flooding effectively (Stark et al. 2016). Importantly, as sea level rise continues to shift intertidal zones landward through coastal squeeze (e.g. Jackson & McIlvenny 2011, Torio & Chmura 2015, Luo et al. 2018), ecoengineering approaches utilising intertidal species will also need the necessary space (and capacity) for the species to retreat (Bilkovic et al. 2016). Where the tidal range is small, or the site is the result of land reclamation, there may be little intertidal area for ecoengineering approaches utilising littoral vegetation (Strain et al. 2017b), necessitating the use of subtidal species (such as shellfish, macroalgae or coral reefs) for shoreline stabilisation (Temmerman et al. 2013).

Environmentally sensitive hard defences

Where hard ecoengineering approaches are deemed most appropriate, several factors typically influence the specifics of their design. The options for environmentally sensitive hard defences will depend on whether the structure is being designed de novo or whether an enhancement of an existing structure is planned.

New structures should be designed to maximise the intertidal surface area available for colonisation and to include habitat structural complexity (e.g. introducing microhabitats) that increase diversity (Loke et al. 2014, 2017) and provide refuges for target species (or their prey resources) from predators and environmental stressors (Moschella et al. 2005, Chapman & Underwood 2011, Loke et al. 2015). A greater and more heterogeneous intertidal area may be achieved by stepping structures (e.g. Barangaroo, in Sydney, Australia), sloping structures (i.e. a 45° angle as opposed to 90°), adding macroscale habitat features such as horizontal fins (e.g. Seattle seawall; for more information, see the section entitled 'Case studies and scaling up', later in this review) or by integrating holes of varying sizes as shelter for fish (Sella & Perkol-Finkel 2015) or invertebrates (Martins et al. 2010, Witt et al. 2012). Porous structures not only are advantageous from an engineering perspective in dampening waves (Burcharth et al. 2007), but also provide much habitat for colonisation, even if much of the additional biodiversity is hidden (Sherrard et al. 2016).

Microhabitats can be incorporated in structures by utilising heterogeneous revetment boulders rather than smooth panelled seawalls or by casting pits, grooves, water-retaining structures and other features into the concrete panel or block (Chapman & Blockley 2009, Chapman & Underwood 2011, Firth et al. 2014, 2016a, Loke et al. 2016, Strain et al. 2018b). Where possible, new structures should utilise materials that encourage the recruitment of target organisms. Recruitment will be dependent on the surface chemistry (Pomerat & Weiss 1947, Harlin & Lindbergh 1977, Anderson 1996, Neo et al. 2009) and roughness (microtopography, Köhler et al. 1999, Nandakumar et al. 2003, Coombes et al. 2015) of the structure. The colour of structures may also influence colonisation (James & Underwood 1994, Satheesh & Wesley 2010) by affecting brightness (many larvae are negatively phototactic; Bayne 1964, Raimondi & Keough 1990) and temperature (dark colours absorb while light colours reflect heat, Kordas et al. 2015, McAfee et al. 2017), although this has received little research attention. Use of local quarried rock may provide a substratum that is physically and chemically most similar to natural rocky shores (Green et al. 2012). Alternatively, the chemical composition of concrete may be altered to encourage recruitment (Sella & Perkol-Finkel 2015, McManus et al. 2018).

Options for existing structures will be more limited. They may include subtractive approaches where microhabitats and area are added by drilling holes, pits and grooves in structures (Chapman & Underwood 2011) or coring larger rock pools (Evans et al. 2016). Alternatively, additive approaches include retrofitting artificial structures with units such as complex tiles (Goff 2010, Loke & Todd 2016, Strain et al. 2017a) and water-retaining flowerpots and artificial rock pools (Chapman & Blockley 2009, Browne & Chapman 2011, Chapman & Underwood 2011, Firth et al. 2014, 2016a, Evans et al. 2016, Morris et al. 2017), which may be constructed from bioenhancing concrete materials (Sella & Perkol-Finkel 2015) or the transplantation of living, habitat-forming species (Perkol-Finkel et al. 2012, Ng et al. 2015), see review by Strain et al. (2018b).

Additionally, it may be possible to add new habitats to existing seawalls by placing boulders at the toe of each seawall (Green et al. 2012, Liversage & Chapman 2018). This also minimises scouring of the main structure by wave driven sand, gravel and cobbles (Moschella et al. 2005). Although some ecoengineered shorelines have been shown to reduce the proportion of nonnative species compared with traditional artificial structures (Sella & Perkol-Finkel 2015), an important aspect of monitoring is to ensure that the ecoengineered habitats are not favouring the establishment or spread of nonnative species (e.g. Morris et al. 2018a). Strategic choices for ecoengineered shoreline design and management surrounding invasion prevention are detailed by Dafforn (2017) and include techniques such as preseeding, managing for strong native grazer and predator populations and modifying the chemical and physical properties of ecoengineering features to facilitate the establishment of native assemblages.

The specific design of environmentally sensitive hard defences should be informed by socioeconomic and ecological goals (see the section 'Evaluating the ecoengineered shorelines approach in practice', later in this review). For example, if there is an ecological goal of biodiversity enhancement, it might be important to incorporate a diversity of microhabitats into structures, so as to maximise the number of niches (Strain et al. 2018b) and enhance beta diversity (cf. Kelly et al. 2016). In contrast, if the goal is enhancing water quality through filtration, microhabitats and chemistry that favour bivalves (or other filter feeders) should be targeted (see Strain et al. 2017a for an example of protective microhabitats that enhance bivalve survival). Morris et al. (2018c) provide a framework for deciding what interventions to apply in order to target particular functional groups of fish. Design interventions can be targeted to enhance exploited species, such as limpets in the Azores (Martins et al. 2010). Enhancing grazers will lead to less slippery and dangerous seaweeds (Jonsson et al. 2006) in areas where structures abut beaches used for recreation.

Soft approaches and the selection of habitat-forming species

Soft ecologically engineered shorelines are typically founded on habitat-forming species. A variety of these species may be suitable for shoreline stabilisation, including (but not limited to) reef-forming invertebrates such as corals (Ferrario et al. 2014), oysters (Scyphers et al. 2011), mussels and worms (Moody 2012), intertidal vegetation such as mangroves and saltmarsh (Kumara et al. 2010, Gittman et al. 2014) or subtidal vegetation, such as seagrass or kelp (Dubi & Tørum 1994, Ondiviela et al. 2014). The choice of habitat-forming species for shoreline protection projects will depend on physicochemical, ecological and socioeconomic factors. First, the selected species must be able to persist, grow and reproduce under both present and projected future environmental conditions, given ongoing climate change and coastal development (Walles et al. 2016). Second, the species must be capable of forming habitat that is able to adequately protect shorelines through stabilisation of sediments or dampening of wave energy under present and projected future environmental conditions (Narayan et al. 2016, Morris et al. 2018b).

Where possible, native species should be selected over nonnative species. However, in rare instances, nonnative species may be more suitable choices due to extirpation of their native functional analogues or their greater capacity than natives to tolerate future environmental change. In the Netherlands, where native flat oysters (*Ostrea edulis*) are now functionally extinct (Beck et al. 2011) and native mussel beds (*Mytilus edulis*) have become largely overgrown by nonnative Pacific oysters, *Crassostrea gigas* (Markert et al. 2010), the nonnative oyster is being utilised as a device for shoreline stabilisation. This is because it is more suited to the present environment and is, to a large degree, functionally equivalent to the native species (da Vriend et al. 2014). Interestingly, an experimental study by Borsje et al. (2011) found that nonnative oyster beds were more effective in wave attenuation than were native mussel beds. Nonnative species, however, should not be introduced for shoreline protection purposes if they are not already naturalised in the targeted environment, or if their presence will lead to negative ecological or socioeconomic impacts that outweigh their benefits (Bunting & Coleman 2014). Ultimately, the habitat-forming species need to be self-sustaining. The probability of this may be maximised where there are nearby patches of extant natural habitat to which the population being used for ecoengineering is connected.

Socioeconomic considerations should also be taken into account when selecting species. For example, shellfish reefs may be a less desirable alternative for ecoengineering in areas subject to high human recreational activity due to risk of cuts and injury (and potential infection) from razor-sharp shells (O'Donnell 2016). Similarly, mangroves may be a socially unacceptable alternative if they facilitate the proliferation of mosquito populations, especially in areas where they are carriers of disease (Temmerman et al. 2013, Friess 2016). Conversely, certain species can be particularly desirable because they provide valued ecosystem goods or services. For instance, in areas where improvement of water quality is a key goal, living shorelines based on shellfish may not only serve to stabilise shorelines, but also improve water clarity (Allen et al. 1992, Wilkinson et al. 1996, Coen et al. 2007, Grabowski & Peterson 2007). Where the habitat-forming species provides food or raw materials, it may need to be protected from harvest so as not to compromise its role in protecting the shoreline (O'Donnell 2016). For example, the success of oyster reef restoration projects in the United States, including those for shoreline protection, is enhanced by their protection within no-take reserves (Powers et al. 2009). In Southeast Asia, firewood collection may compromise the long-term persistence of mangrove forests unless appropriate protective measures are put in place (Malik et al. 2015). Another notable socioeconomic goal can be achieved through habitat-forming species that secrete calcareous skeletons, such as oysters, corals and tube worms, through bioprotection of underlying rock or concrete, which potentially increases strength and longevity of structures (Coombes et al. 2013, 2017, Coombes, 2014).

Where habitat-forming species are being established from a baseline of zero at the shoreline site and the shoreline is already under threat from erosion or inundation, temporary protective measures may need to be in place while the habitat establishes. An example of this is the use of cultivator pots, protective matting and structures for mangrove seedlings (Krumholz & Jadot 2009, Tamin et al. 2011). These can help shield the habitat-forming species from erosion and stimulate establishment through enhanced sediment accretion, as well as protecting the shoreline from erosion until the habitat-forming species is able to attain sufficient size and/or density to fulfil the shoreline protection function. Knowledge of the configurations and morphologies of the habitat-forming species that best serve the functional role of shoreline protection will assist in determining which populations of the habitat-forming species to transplant to the site and the optimal way in which transplantation should proceed in order to achieve the desired goal. In some instances, transplantation of multiple habitat-forming species that exist in adjacent or nested configurations may best stabilise the shoreline and offer enhanced ecological cobenefits where they support unique species assemblages or promote distinct ecosystem functions (Gribben et al. this issue).

Hybrid approaches

Hybrid approaches combine hard defences with natural elements, such as restoring key species or habitats by plantings and seeding. Such approaches can provide considerable benefits by helping to support broader management and conservation goals, enhancing ecosystem services and/or providing added protection beyond that of traditional hard structures (O'Donnell 2016). They enable ecological communities to be incorporated into defences at sites where they would not survive on their own or would, on their own, provide inadequate coastal protection (Sutton-Grier et al. 2015).

The configuration of natural and hard elements in a hybrid approach depends on the wave exposure of the environment, the amount of space available for transplants and the species involved. At sites where natural habitats persist and provide shoreline protection from moderate (but not large) storms, they may be fringed on their landward side by hard defences, such as seawalls or revetments, which provide the required protection from major storm events. At sites where wave action inhibits establishment of unprotected living habitat, the hard defence may be placed on the seaward side to protect the habitat from erosion. Low-crested rock sills are used to stabilise landward sediments in many instances, so that saltmarsh (Benoit et al. 2007, Bilkovic & Mitchell 2013) or mangroves (Hashim et al. 2010) can establish and grow. Hard defences and living habitat can also be interspersed. In Florida, for example, mangroves were planted in concrete cultivars that attenuate waves, accrete sediment and are designed to provide more favourable environments for survival and growth (Krumholz & Jadot 2009). This technique is also currently being implemented in Victoria, Australia (https://www.climatechange.vic.gov.au/__data/assets/pdf_file/0018/123507/Climate-Change-Innovation-Grants_funded-projects.pdf).

Generally, as wave energy increases, a greater ratio of hard defence to soft defence via a living habitat will be required. In environments with limited intertidal space, the opportunity for living habitat may be limited to small habitat pockets among hard defences (see the pocket beach example in Seattle, described in Toft et al. 2013). Nevertheless, the inclusion of these can still lead to significant biodiversity enhancement and ecosystem service provision over those provided by the hard defences alone (Spalding et al. 2013).

Implementation of ecoengineered shorelines

The implementation of the ecoengineered shoreline approach is a complex process that is influenced by politics, public safety, community values and cost. Thus, it is not without potential risks and challenges. A major concern is dealing with communicating uncertainties and the need for adequate data (Chapman & Underwood 2011). Poor understanding of the underpinning mechanisms of common ecoengineering techniques (Gedan et al. 2011, Bouma et al. 2014) can limit the effectiveness of ecoengineered shoreline designs and lead to unanticipated changes in important ecosystem processes (Bilkovic & Mitchell 2013). There may be unintended consequences of standard ecoengineering strategies, such as enhanced structural complexity that may lead to increased accumulation of litter by shoreline structures (Aguilera et al. 2016). There is a lack of accepted and standardised criteria for evaluating their success (Perkins et al. 2015). A reputational risk is the misuse of ecoengineered shoreline concepts and terminology to justify and market expanded loss and conversion of natural habitats as ecologically desirable (Pilkey et al. 2012). However, the ecoengineered shoreline approach also has numerous potential benefits, which have already been discussed in this review, and which have been strongly emphasised in the recent research literature (Pioch et al. 2018).

The concept of working with nature has existed for centuries, but it was little less than half a century ago when the term *ecoengineering* was framed and the concept recognised (see the section entitled 'Introduction and the history of ecological engineering of shorelines', earlier in this review). Although older work on the ecology of artificial habitats exists (e.g. Plymouth and Port Erin breakwaters, United Kingdom; Southward & Orton 1954, Kain & Jones 1975) and a broader interest in the ecology (Hawkins & Cashmore 1993) and restoration of artificial marine habitats emerged in the 1980s and 1990s (e.g. Russell et al. 1983, Allen et al. 1992, 1995, Hawkins et al. 1992a,b, 1993, 1999, 2002, Allen & Hawkins 1993), research on ecoengineering of coastal structures only began to accelerate in the early 2000s (Chapman & Underwood 2011, Perkins et al. 2015, Strain et al. 2018b). In part, this increase has been prompted by growing recognition of the benefits of the ecoengineered shoreline approach (Pioch et al. 2018) the importance and relevance of which are particularly evident when considered within the context of major ecological, social and economic challenges currently faced by coastal ecosystems and human communities. It is within this context that we identify several drivers of the recent growing interest in ecoengineered shorelines, which we have broadly classified into five categories (discussed in the next sections): (1) climate change and associated sea level rise and increased flood risk; (2) coastal urbanisation, economic development and urban renewal; (3) integration in governance and policy; (4) growing public outreach and awareness and (5) advances in science and technology.

Climate change, the associated sea level rise and increased flood risk

Anthropogenic climate change is placing coastal communities at risk; consequently, adaptation strategies are likely to be urgently required, especially in low-elevation coastal areas (Neumann et al. 2015, Dangendorf et al. 2017). Rising sea levels and increasingly frequent or more intense storms are leading to coastal erosion and flooding, necessitating either relocation or defence of low-lying settlements and infrastructure. By the end of this century, a rise of at least 0.25–0.55 m is projected, and about 95% of the ocean area will be affected (IPCC 2014). It is estimated that in the United States, approximately 1500 homes will be lost to coastal erosion each year for several decades, at a cost to property owners of $530 million per year (Heinz Centre 2000). These impacts of climate change are exacerbated by continued population growth in the coastal zone (Nicholls et al. 1999). Worldwide, it is predicted that by 2070, 37 million people and assets worth $13 trillion will be exposed to coastal hazards such as storms, flooding and climate variability (Nicholls & Hanson 2013). With carbon emissions still not stabilised globally, development of climate change adaptation strategies is now a priority for many jurisdictions (IPCC 2014).

Disasters such as the Indian Ocean tsunami in December 2004 and the hurricanes Katrina (2005), Sandy (2012) and Michael (2018) in the United States have drawn attention to the important role that coastal vegetation and reef-forming invertebrates play in shielding populations and property from sea level rise and storms (e.g. Danielsen et al. 2005, Gedan et al. 2011, Arkema et al. 2013). In some instances, the role that natural ecosystems play in reducing flooding is more effective than that of hard defences (Gittman et al. 2014). For example, along the east coast of the United States, damage to shorelines protected by marsh was less than along shoreline protected by bulkheads following category 1 hurricane events (Gittman et al. 2014). In several countries, including the United States, these observations, coupled with the growing need for coastal settlements and infrastructure to adapt to the threat of climate change, has led to policy that focusses attention away from hard to soft (i.e. 'living shoreline') defences as a climate change adaptational strategy (Finucane et al. 2014).

Despite the catastrophic effects that natural disasters may have on urban settlements, they can also bring about changes in public sentiment and government action. For example, Hurricane Sandy caused extensive damage to more than 650,000 homes along the east coast of the United States and claimed over 150 lives (Stewart, 2014). In response, the U.S. government provided more than $50 billion towards disaster relief and established the Hurricane Sandy Rebuilding Task Force to examine how funds might be best used to improve the resilience of communities and infrastructure to future and existing threats, including those exacerbated by climate change. The resulting Infrastructure Resilience Guidelines (Finucane et al. 2014) called for 'environmentally sustainable and innovative solutions that consider natural infrastructure options in all Federal Sandy infrastructure investments'. As a result, large numbers of living shoreline projects have now been implemented across the coastline (e.g. Tom's Cove and Assateague Beach), which are anticipated to provide not only coastal defence, but also other societal benefits.

Coastal urbanisation, economic development and urban renewal

As the human population living in the coastal zone continues to increase, rapid urbanisation is transforming natural shorelines, expanding the built environment and increasing land reclamation. Developed urban shorelines are characteristically dominated by artificial structures, which have finite service life spans and need to be maintained regularly and replaced eventually (Mahmoodian et al. 2015). Urban renewal provides opportunities for converting existing coastal defences into more ecologically sustainable designs. For example, the Seattle seawall (see the section entitled 'Case studies and scaling up', later in this review) was failing and at risk of collapse should an earthquake hit, posing a public safety hazard. This, combined with the importance of the shoreline as a migratory corridor for endangered juvenile Chinook salmon, provided the opportunity to incorporate habitat improvements to the seawall reconstruction. Ecoengineered shoreline approaches will be adopted most readily where they provide clear benefits, which are in the public interest.

It is not unusual for areas of a city to be intentionally or unintentionally deindustrialised or deurbanised back to a more natural situation, and in such cases, ecoengineered shorelines can potentially play an important role. The Barangaroo Reserve in Sydney, Australia, is an example of a project driven by deindustrialisation and redevelopment. This reserve was once a concrete container terminal, but it underwent a major transformation to become a harbour foreshore park. Projects of this sort have great potential and open up huge opportunities for ecoengineered shorelines. Obsolete, locked dock basins in macrotidal estuaries in Europe have been the focus of ambitious urban renewal schemes, usually retaining the dock basins as water features or marinas and turning warehouses into housing, offices, hotels, shops, museums, art galleries and entertainment venues. One of the best studied has been the Liverpool dock system, where water quality was improved by artificial mixing and biofiltration by artificially (Russell et al. 1983) and naturally settled mussels (Allen et al. 1992, Hawkins et al. 1992a, Allen & Hawkins 1993, Wilkinson et al. 1996), leading to increases in biodiversity and healthy inner-city ecosystems of considerable conservation interest (Allen et al. 1995). The docks were much used in outreach and public engagement (Hawkins et al. 1993). Had the ecoengineered shoreline concept existed in the 1980s, considerable impetus and a body of good practice could have been deployed at the planning stage of urban renewal in Liverpool.

Integration into governance and policy

Political will and shifts in governance and policy represent a strong and effective driver of ecoengineered shoreline development (Arkema et al. 2017, Sutton-Grier et al. 2018). For example, the central government of China issued a document on the use and protection of the country's coast in 2017, which specified that at least 35% of its shoreline should be maintained in a natural state by 2020 and that all proposed reclamation projects should be halted. This new national policy has sparked interest among provincial governments to invest in ecoengineered shoreline development and to convert concrete shorelines into more environmentally friendly ones by means of ecoengineering (Duan et al. 2016). At a state level, Maryland in the United States introduced the Living Shorelines Protection Act in 2008, which requires that property owners use living shorelines as the default for protection unless the owner can demonstrate the need for an artificial structure (e.g. a bulkhead) instead (Kochnower et al. 2015). Following up on European Union Directives and policy guidelines, the UK government and its devolved administrations (e.g. Wales) have given considerable emphasis on forming policy to incorporate habitat restoration and using ecoengineering in new coastal developments, including coastal defences in emerging planning guidance (see Evans et al. 2017, 2019 for detailed reviews of these efforts).

Global concerns for climate change and maintenance of biodiversity as seen in the participation of different governments in international agreements and conventions may also lead to ecoengineering options. The Paris Climate Agreement states that signatory parties 'should take action to conserve and enhance, as appropriate, sinks and reservoirs of greenhouse gases' (United Nations 2015), and the Convention on Biodiversity Conservation aims to achieve 'the conservation of biological diversity, the sustainable use of its components' (United Nations 1992). Ecoengineering is potentially one of the means that can help achieve the goals of these agreements and conventions.

Growing public outreach and awareness

Public awareness of and concern about marine environmental issues are positively correlated with being well informed (Gelcich et al. 2014). Recent expanded interest in ecoengineered shorelines thus may be tied to greater education and understanding of the topic, as well as greater experience with coastal hazards that necessitate coastal defence infrastructure (Kochnower et al. 2015, Gray et al. 2017). Interviews with coastal residents in the United States suggest that awareness of and preference for ecoengineered shoreline approaches over traditional engineering are common, although so too are various misconceptions about the topic (Scyphers et al. 2014). In a recent survey of residents of four harbours in Australia and New Zealand, Kienker et al. (2018) found that level of education was positively correlated with support for marine ecoengineering. Additionally, prior knowledge about the dominant marine artificial structures in their harbour influenced how much participants in the study were willing to pay for ecoengineering (Kienker et al. 2018). A study of a variety of different stakeholders in Wales, ranging from engineers through coastal ecologists to the general public highlighted a willingness to embrace a multifunctional approach to the design of sea defences, especially if ecological objectives could also be delivered (Evans et al. 2017).

Education may result from formal programmes, such as in schools, or from community engagement in outreach activities and media articles (Figure 4). Emphasis on raising public awareness has been central to several recent ecoengineered shoreline projects. For example, the Billion Oyster Project in New York integrated curriculum development and restoration-based education programmes about the importance of oyster reefs and their ecosystem services for primary and high school students (Janis et al. 2016), which likely contributed to public awareness of the project. Local media attention has also recently increased the visibility of various ecoengineered shoreline projects, such as flowerpots on Sydney seawalls (e.g. Chapman & Blockley 2009, Browne & Chapman 2011, Morris et al. 2017).

Advances in science and technology

Over the past two decades, there has been rapid growth in the understanding of how shoreline defences affect coastal ecosystems (e.g. loss of biodiversity, increase in invasive species) (Airoldi et al. 2005a, Martin et al. 2005, Bulleri & Chapman 2010), and how these negative effects may be compensated for through ecoengineered shoreline designs (Moschella et al. 2005, Perkins et al. 2015, Firth et al. 2016a, Munsch et al. 2017b, Strain et al. 2018b). It is now acknowledged that some artificial defences can cause new erosion problems or loss of public amenities (e.g. beach loss in front of seawalls; Fletcher et al. 1997). Recent research has increasingly emphasised ecoengineered shorelines as a potentially more cost-effective strategy than traditional approaches to coastal protection (Sutton-Grier et al. 2018). As studies on ecoengineered shorelines have multiplied, the accumulation of data and research findings have enabled meta-analyses pinpointing the precise ecoengineering strategies and specific circumstances under which ecoengineered shorelines are likely to be successful at meeting their stated aims (Gedan et al. 2011, Bugnot et al. 2018, Strain et al. 2018b).

Figure 4 Various signs at ecoengineered shorelines to raise awareness and public education.

In addition, many ecoengineered shoreline strategies have themselves advanced and become more cost effective because of recent innovations in fabrication (3-dimensional printing) and materials science (e.g. ecofriendly cement mixes; Perkol-Finkel & Sella 2014). Further, advances in ecology and ecosystem science have brought insights that clarify ecological processes and the functional traits of taxa that are central to ecosystem services provided by shoreline habitats. For example, greater understanding of the role of oysters in biofiltration, habitat provisioning, sediment stabilisation and wave attenuation in New York Harbour (e.g. Coen et al. 2007, Grabowski et al. 2012) ultimately facilitates ecoengineered shoreline projects that are of ecological and social value (e.g. the Billion Oysters Project). As knowledge of coastal ecological processes and the science of ecoengineering continue to grow and support advances in technology, it is likely to drive the demand for ecoengineered shorelines.

Barriers to implementation

In general, barriers may include (1) a lack of awareness by decision-makers of the available options and a failure to engage with the multiple stakeholders involved in communicating the benefits of ecoengineered shoreline approaches, (2) government policies that are insufficiently flexible to accommodate new approaches and ways of thinking and (3) an absence or paucity of science-based evidence for the success, benefits and long-term durability of innovations (see the section entitled

'Evaluating the ecoengineered shoreline approach in practice', later in this review). Additionally, there may be cultural, financial or logistic (e.g. extreme environmental degradation) barriers to adopting ecoengineered shoreline approaches. To incentivise the innovation and development of ecoengineered shoreline designs and technologies, a consistent and supportive framework is required to improve risk-to-reward ratios, particularly during the demonstration stage of technology and design development. This framework may enhance positive expectations, stimulate learning, improve the design and increase the likelihood of successful project implementation (Foxon et al. 2005).

Awareness and communication among key stakeholders

For hard ecoengineered shorelines, small-scale trials have been carried out in different parts of the world over the last three decades, but medium- to large-scale implementation of ecoengineered shorelines is still in its infancy (but see the Seattle case study as one exception). For soft engineering, larger-scale implementation is more common; for instance, there has been a great effort to restore mangroves in Asia, with coastal defence as a driver in many projects (Saenger & Siddiqi 1993, Benthem et al. 1999, IFRC 2011). However, in contrast to hard ecoengineering, which has often been research driven, these larger-scale soft ecoengineering projects frequently lack scientific rigour and robust ecological-monitoring regimes (Narayan et al. 2016). Despite hot spots of research and/or implementation of ecoengineered shorelines globally, one barrier to their implementation more generally may be the wider transfer of information to relevant local, state or national governments. Even in countries that are driving ecoengineered shoreline research (like Australia, European countries, and the United States), regional (state or provincial) governments may not know about the rationale, concept and potential value of ecoengineered shoreline implementation. A better understanding of these innovations by government leaders, key stakeholders and the general public may lead to support for small-scale project trials. In turn, the success of these trials can lead to the implementation of large-scale ecoengineered shoreline projects.

Effective communication and partnerships among architects, engineers, ecologists, socioeconomic scientists and other relevant stakeholders are also important. Cross-disciplinary teams generate holistic projects through collaboration to achieve a common goal and build synergy. In particular, there is a need to build trust and better understanding among stakeholders, who often do not share the same perspective and interests (Prati et al. 2016), so as to develop ecoengineered shorelines that can fulfil both engineering requirements (i.e. shore protection and structure integrity) and ecological goals, while also providing a social benefit (Evans et al. 2017).

A better international strategy for enhancing the awareness of ecoengineered shorelines and their benefits to people and the environment is needed. The closest international tool to address this need currently is the development of an online coastal resilience mapping interface, driven by The Nature Conservancy in the United States, which has attempted to synthesise data on the coastal protection provided by intact natural habitats and to identify soft engineering projects (http://www.maps.coastalresilience.org/cities/). This tool does not include hard ecoengineered shorelines. The National Oceanic and Atmospheric Administration (NOAA) has developed a useful webpage to introduce the concept of living shorelines, with some examples (https://www.fisheries.noaa.gov/insight/living-shorelines) in the United States. In Australia, the state government of New South Wales has developed an Environmentally Friendly Seawalls guide (http://www.environment.nsw.gov.au/resources/estuaries/pubs/090328-Seawall-Guide-2012-Reprint.pdf), which summarises local projects.

Ideally, an internationally oriented ecoengineered shoreline organisation and educational website should be established, accompanied by a campaign aimed at increasing awareness at a global level. This effort could capitalise on existing networks, such as Restore America's Estuaries and the World Harbour Project, and include international organisations such as The Nature Conservancy and World Wide Fund for Nature to enhance content delivery. The strategies could include using multimedia, both conventional and Internet-based, to introduce ecoengineered shorelines and highlight illustrative projects from around the world.

Policies, incentives, regulations and financial instruments

At present, specific policies that call for integrating ecological considerations into the design and construction of coastal defence and coastal development schemes through ecological engineering are generally lacking in most countries or jurisdictions. Nevertheless, ecoengineered shorelines can potentially make a significant contribution towards a wide variety of current policy objectives, such as those that support sustainable development and the maintenance of biodiversity (Naylor et al. 2012, Dafforn et al. 2015b). For example, under the United Nations Convention on the Law of the Sea (UNCLOS), states are required to protect and preserve the marine environment (Ban et al. 2014). In this context, ecoengineered shorelines could be proposed as a tool for generating more adaptive, resilient coastlines for the mutual benefit of the environment and society. Having ecoengineered shorelines on the international policy agenda may incentivise the implementation of local legislation to further support their application (e.g. the Living Shorelines Act, Maryland).

The few large-scale ecoengineered shoreline projects emerging recently (e.g. Tong King Delta and Mekong Delta, Vietnam; Grenville Bay, Grenada; Barangaroo Reserve, Carss Park and Olympic Park in Sydney; the Seattle Seawall Project, Harlem River Designing the Edge Project, Brooklyn Bridge Park, and Vancouver Conference Centre Project in the USA), if monitored efficiently for successes and failures, could provide valuable evidence to inform the adoption of ecoengineered shoreline designs, but also offer some essential information for doing a benefit-cost analysis (BCA) of any proposed ecoengineered shoreline project. Insufficient resources are usually the main barrier for any innovative development and its commercialisation (Hadjimanolis 1999). This situation is exacerbated by the lack of full accounting of the benefits associated with ecoengineered shorelines, which has rarely been performed. This is due primarily to the difficulty in assigning monetary values to the ecosystems and social services that the projects are designed to perform (see the section entitled 'Measuring socioeconomic outcomes', later in this review; but see Narayan et al. 2016, Reguero et al. 2018).

To date, many of the documented ecoengineered shoreline projects have been on a small scale, partly due to insufficient financial support. In most cases, local governments play a key role by providing some seed funding for conducting feasibility studies on small-scale trials of various ecoengineered shoreline designs. If a trial project proves successful, the government might proceed to conduct a large-scale implementation, depending on the policy agenda, site availability and financial resources. Unfortunately, even when money is available, in some cases the type of monitoring that is performed focusses only on proving whether a particular trial concept can or should be scaled up at a particular site (J. Miller, pers. obs.). In these cases, the opportunity to identify benefits that may apply more broadly to ecoengineered shorelines constructed in other locations is lost.

Certain countries have structured regulations that may encourage implementation of ecoengineered shoreline designs. For example, in the United States, there are regulations that demand quantitative mitigation measures for development projects, including in coastal and marine environments. These typically call for the restoration of habitat loss and/or ecosystem functions affected by the project and are often significantly higher in cases where on-site mitigation is not possible in terms of the investment expected (e.g. Ambrose 1994). Currently, mitigation measures include actual restoration work (e.g. marsh restoration, construction of artificial reefs) or purchasing mitigation credits from available mitigation banks. Ecoengineered shoreline implementation can be viewed as on-site mitigation measures, potentially offsetting some of the overall compensation and mitigation costs. This can effectively incentivise the implementation of ecoengineered shorelines; however, documentation of such cases is lacking within the United States, and currently this approach is not widely adopted in other countries.

For green buildings, there are certification and grading systems, such as those by the Leadership in Energy and Environmental Design (LEED) initiated in the United States (Humbert et al. 2006) and nowadays applied in many countries worldwide, the Building Research Establishment Environmental Assessment Method (BREEAM) in the United Kingdom (Crawley & Aho 1999) and Building

Environmental Assessment Methods (BEAM) in Hong Kong (Lee et al. 2007) in order to recognise the environmental friendliness and energy-saving efforts being adopted in the building as an incentive for the project proponent or developer to achieve the highest rating (e.g. platinum award in LEED).

Recently, the Waterfront Edge Design Guidelines (WEDG) in New York and New Jersey Harbour have been developed by the Waterfront Alliance (http://waterfrontalliance.org). The WEDG is a point-based rating system with a set of guidelines for waterfront projects in New York and New Jersey Harbour, with a view to creating resilient, ecological and accessible waterfronts (Box 1). During the evaluation process, actions taken in the planning, design, construction and even postconstruction (monitoring) process are tallied to determine how successful a project is in achieving benefits in the three core areas: resilience, ecology and access. Any project proponents can voluntarily go through the WEDG assessment to gain certification and receive advice from professional assessors to further improve the project in the plan's three aspects: shoreline protection, ecological enhancement and diverse uses of the waterfront by various stakeholders.

Thus far, the WEDG has been used to certify over a half-dozen projects spanning a range of shoreline uses, from parks to heavy industrial areas in New York City (http://wedg.waterfrontalliance.org). Recently, the WEDG has undergone a revision intended to streamline the ratings system and make it more readily applicable to shorelines outside the New York/New Jersey metropolitan

BOX 1 THE WEDG CERTIFICATION SYSTEM FOR WATERFRONT PROJECTS ADOPTED IN NEW YORK AND NEW JERSEY

WEDG

The WEDG employs an evidence-based approach, focussing on three key pillars of excellent waterfront design in New York and New Jersey Harbour. They include the following:

- *Resilience*: The waterfront project should reduce risks or be adaptable to the effects of sea level rise and increased coastal flooding, through setbacks, structural protection and other integrative landscaping measures.
- *Ecology*: The waterfront project should protect existing aquatic habitats and use designs, materials and shoreline configurations to improve the ecological function of the coastal zone and strive to be consistent with regional ecological goals.
- *Access*: The waterfront project should be equitable and informed by the community, enhancing public access, supporting a diversity of uses, from maritime, recreation and commerce where appropriate, thereby maximising the diversity of the harbour and waterfront.

The credit points made by the professional assessors would help guide the design process, from conceptual design through operations, and provide design performance goals for resilience, ecology and access in the following six categories:

- Category 0: Site Assessment and Planning
- Category 1: Responsible Siting and Coastal Risk Reduction
- Category 2: Community Access and Connections
- Category 3: Edge Resilience
- Category 4: Natural Resources
- Category 5: Innovation

Source: http://wedg.waterfrontalliance.org/

area. The WEDG is the first example of this kind to pave the way for developing an international certification system that will play a key role in incentivising the development of ecoengineered shorelines around the globe. Furthermore, the question is how to use the certification system to further incentivise ecoengineered shorelines. Potentially, if waterfront projects have successfully addressed the Sustainable Goals of the United Nations (https://www.un.org/sustainabledevelopment/sustainable-development-goals) and received certification by a system like the WEDG, then the project proponent could receive a tax reduction or rebate and banks could provide better loans with a lower interest rate for the project proponent. However, such arrangements are likely to depend on individual governments and banks. There is certainly a need to build more successful precedent cases to convince governments and financial institutions.

Engineering concerns and paucity of science-based evidence for success

One of the biggest obstacles from an engineering standpoint is liability (Slate et al. 2007). Although regulations differ from country to country, engineers are generally responsible for ensuring public safety and can be held responsible if their projects fail. Findings of fault can result in severe economic, professional or even criminal sanctions (Baura 2006). As a result, engineers tend to be cautious and view unproven technologies/approaches with an appropriate degree of trepidation. When it comes to the design of traditional infrastructure such as buildings and bridges, engineers can rely on established codes such as the International Building Code for guidance.

For less traditional infrastructures such as coastal defence systems (i.e. seawalls, breakwaters and revetments), codes are often lacking. As a result, engineering design is often guided by a set of standard practices/approaches advocated in respected design manuals such as the *Coastal Engineering Manual* (U.S. Army Corps of Engineers 2002) and *The Rock Manual* (CIRIA et al. 2007). Absent codes, following well-vetted engineering practices provides a level of protection from professional liability should a project fail. For the engineering community to more openly embrace ecoengineered shorelines, design manuals need to be developed (Burcharth et al. 2007), or perhaps more likely, sections need to be added to existing design manuals to deal specifically with ecoengineered shoreline approaches. Such an effort needs to be led by an internationally recognised body such as the U.S. Army Corps of Engineers or Construction Industry Research and Information Association (CIRIA).

A second, related concern from an engineering standpoint is the real or perceived lack of predictability of the behaviour of certain types of ecoengineered shoreline projects with time (Bouma et al. 2014). Ecoengineered shorelines have a temporal component that is more difficult to deal with than traditional engineering materials. For example, fatigue is a well-known engineering phenomenon in which materials lose strength over time due to repetitive loading. Most traditional engineering materials have gone through enough testing that this fatigue behaviour can be predicted and planned for during design. The living element of ecoengineered shoreline projects is much less predictable (and certainly much less well studied) than the temporal behaviour of traditional materials (Bouma et al. 2014). Furthermore, some ecoengineered shoreline approaches may actually depend on the living portion of the project for properties critical to the performance of the structure. For example, if oyster growth is critical to the wave attenuation or chloride penetration prevention characteristics of a project, the threat of environmental phenomena affecting growth has a potentially devastating, cascading impact on the engineering characteristics. This can be related back to liability and the reluctance of engineers to design something whose behaviour they cannot predict. A robust evidence base on the efficacy of ecoengineered shoreline approaches was identified by some stakeholders, however, as a barrier to fuller implementation (Evans et al. 2017). Part of the problem is that the evidence is accumulating, but it is not necessarily being delivered in a readily accessible and digestible form to key practitioners, statutory bodies and legislators.

Evaluating the ecoengineered shoreline approach in practice

Traditional artificial defence structures are constructed to protect land from erosion and flooding. Their success is typically measured from an engineering standpoint of whether protection was achieved or not. In contrast, ecoengineered shorelines are valued for their potential to provide multifunctional benefits, and thus measuring their success calls for a wider analysis. We suggest three main domains of objectives for which the success of ecoengineered shorelines may be measured:

- Ecological objectives, often stated in terms of enhancement of biodiversity or a particular species (e.g. salmon in Seattle; Munsch et al. 2017b) or the reestablishment of a habitat that provides ecosystem services such as water filtration or biodiversity provision (e.g. oyster reefs)
- Engineering objectives, which relate to the function of an engineered insertion over its projected life span
- Socioeconomic objectives, such as public acceptance, politics, recreational value, property prices and development goals (Table 1)

Table 1 Examples of criteria for measuring the success of ecoengineered shorelines

Category	Goals: Functions/services	Measure	Controls/reference
Ecology	Native species biodiversity	At species level: species richness, biomass, abundance, percentage of cover, percentage, community assemblage At habitat level: Habitat diversity At genetic level: Genetic diversity	Artificial structure Natural shoreline Before-after
	Invasive species	Number and abundance of species Ratio of native to invasive species	Artificial structure Natural shoreline Before-after
	Target species	Enhancement/recovery of abundance or survival of target species	Artificial structure Natural shoreline Before-after
	Ecological functioning and processes	Integrity of biological assemblage (e.g. functional groups) Biofiltration Water quality Primary productivity Ecosystem engineers/habitat-forming species Bioprotection Carbon sequestration	Artificial structure Natural shoreline Before-after
	Fisheries production	Enhancement of fisheries supply Usage of habitat/refuge by larvae	Artificial structure Natural shoreline Before-after
	Connectivity	Enhancement and/or reduction of connectivity	Artificial structure Natural shoreline Before-after
Engineering (risk reduction)	Energy attenuation	Wave height reduction Current reduction	Natural shoreline Artificial structure Before-after
	Shoreline stabilisation	Horizontal shoreline location	Natural shoreline Artificial structure Before-after

(*Continued*)

Table 1 (Continued) Examples of criteria for measuring the success of ecoengineered shorelines

Category	Goals: Functions/services	Measure	Controls/reference
	Achieving adequate/desirable sedimentation dynamics	Regulating sediment accumulation	Natural shoreline Artificial structure Before-after
	Reduce flooding	Surge extent/height	Natural shoreline Artificial structure Before-after
	Structural integrity	Structural integrity Durability/longevity Resistance to extreme weather events	Through time Natural shoreline Artificial structure
Social	Aesthetics	Appeal to people	Before-after Areas with and without interventions
	Tourism and recreation	Waterfront accessibility People's awareness and use of the waterfront	Before-after Areas with and without intervention
	Education	People's knowledge and awareness of coastal biodiversity and ecoengineered shorelines	Before-after Areas with and without intervention
Governance and policy	Facilitate use of ecoengineered shorelines	Funding incentives, permits, recommendations, and regulations	Before-after
	Hazard mitigation	Protection of property and life	Before-after Natural shoreline Artificial structure
Economic	Shore protection insurability	Reduction of economic loss due to damage Reduction of insurance premium requirement Increase in property value	Before-after Natural shoreline Artificial structure
	Creation of jobs	Increase in job opportunities in landscape architecture, marine ecology, construction, tourism and fishery sectors	Before-after
	Business opportunity	Nurturing new businesses associated with successful waterfront	Before-after
	Project performance	Construction, maintenance and services provided	Artificial structure

Note: The term *artificial structure* refers to a traditional engineering solution (e.g. seawalls). *Natural shoreline* refers to the intact equivalent of the created habitat (e.g. mangroves, rocky shores). A bare reference without a structure (e.g. mudflats) may also be an appropriate control for some measurements.

Metrics relating to each of these domains are essential for assessing multifunctionality and overall benefits of ecoengineered shorelines. Although a combination of these domains is becoming increasingly common in monitoring plans for ecoengineered shorelines (Yepsen et al. 2016), we show that in practice, few projects have incorporated all three (marine scientists/managers, engineers and the public; see Figure 5).

Measuring ecological outcomes

Ecoengineered shorelines can be designed to meet a variety of ecological objectives, including biodiversity enhancement, invasion resistance, facilitation of specific target species, specific ecosystem

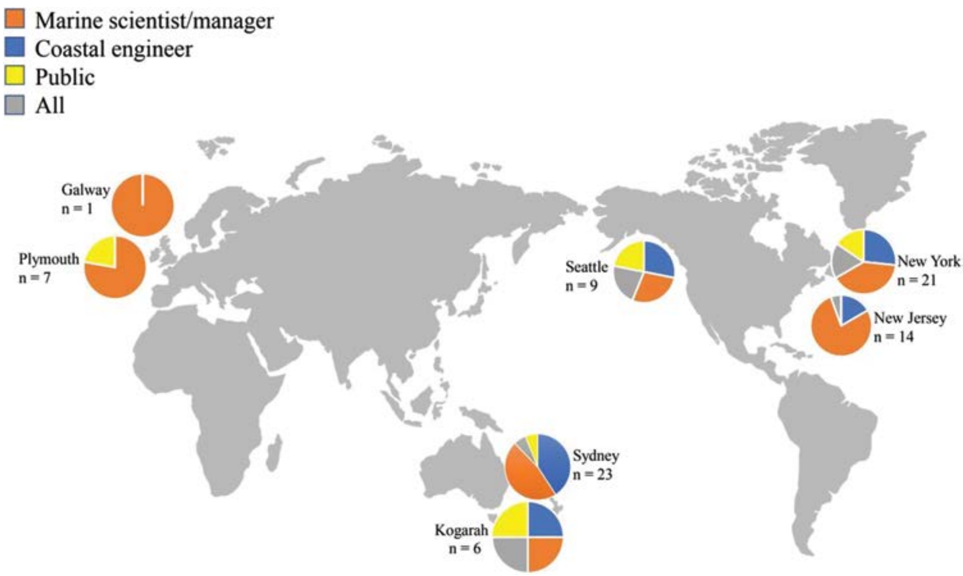

Figure 5 The proportion of key stakeholders (marine scientists/managers, engineers, public and all three) involved in ecoengineered shoreline projects in select locations in the United States, Australia, Europe and the United Kingdom. The data were provided by researchers in these places, and the locations presented were based on where the information about stakeholder participation was available.

processes and general functioning, with the ultimate aim of provision of services (Table 1). The objectives selected depend on the type of ecoengineered shoreline project, the ecological and environmental context in which the project takes place and local societal needs and interests. Ecological objectives for hard ecoengineered shorelines tend to focus on ameliorating abiotic and/or biotic stressors that limit diversity in general, thereby enhancing focal species of conservation or commercial interest or increasing native populations or functional groups that enhance ecosystem services (Chapman & Blockley 2009, Dugan et al. 2011, Perkol-Finkel et al. 2012, Mayer-Pinto et al. 2018, Strain et al. 2018b). In contrast, where soft or hybrid ecoengineered shorelines can be constructed by establishing coastal plants, shellfish or coral reefs (Arkema et al. 2013), the main ecological focus is typically on selecting and monitoring the efficacy of specific habitat-forming organisms for coastal protection via the dampening of waves or stabilisation of sediment (Borsje et al. 2011). However, measures of ecological cobenefits, such as enhancement of biodiversity or fisheries productivity, carbon sequestration and improvement of water quality (Barbier et al. 2011, Davis et al. 2015, Sutton-Grier et al. 2015) may also be important for quantifying the success of soft and hybrid ecoengineered shorelines.

Measuring the ecological benefits of ecoengineered shorelines requires prespecifying a monitoring framework (prior to initiation) that effectively assesses changes in a specific intervention or habitat of interest (Michener 1997, Block et al. 2001). Ideally, monitoring plans compare ecoengineered shorelines with controls (i.e. unaltered artificial shorelines), as well as with ‘reference areas’ (i.e. natural habitats that the ecoengineered shorelines are seeking to mimic). In this way, planned ecoengineered shoreline interventions function as experiments (Chapman et al. 2018). Appropriate controls and reference areas depend on the type of ecoengineered shorelines being constructed. For hard ecoengineered shorelines, the relevant control is typically a new or cleared section (i.e. with any existing flora and fauna removed) of an artificial structure at the same site or at a nearby site with comparable environmental conditions, depending on the scale of the manipulation (Chapman et al. 2018), and reference sites are frequently rocky shores, the closest natural hard-substrate analogues (Chapman & Underwood 2011, Evans et al. 2016).

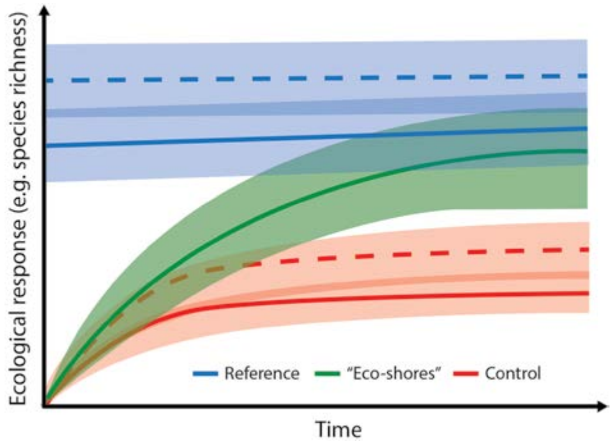

Figure 6 A hypothetical illustration of plausible temporal changes in ecological performance of an ecoengineered shoreline and multiple control and reference sites (solid and dashed lines indicate the various sites). The shaded area around each line represents variation in a single hypothetical site or plot. It is expected that ecoengineered shorelines will enhance the targeted ecological response to be more similar to the reference habitat of interest (e.g. natural habitat), in comparison to the control (i.e. the unaltered structure), which will remain the same.

For soft-engineering interventions, controls should be established patches of the habitat type being constructed. There have also been some attempts to compare soft and hybrid interventions with 'hard' reference sites (e.g. coral reefs with breakwaters and saltmarsh with and without sills with bulkheads; Ferrario et al. 2014, Gittman et al. 2014). However, it is important to note that reference sites may have differences in physiochemical parameters, community succession and other environmental drivers, which can make comparisons between ecoengineered shorelines and reference sites difficult to interpret (Chapman et al. 2018), especially on highly modified coastlines.

Data regarding the ecological performance of reference sites can provide helpful insight into the expected natural variability in recruitment (often due to chance events in plankton) or survival of organisms (e.g. due to heat stress and predation), as well as the possible impacts from other environmental stressors over time (e.g. climate change or fishing; see Figure 6). At least two natural reference sites should be surveyed multiple times both before and after ecoengineered shoreline installation (Beyond BACI; Underwood 1991). For example, if the ecoengineered shoreline is located between two rocky shores, then both rocky shores should be used as reference sites and surveyed regularly alongside the monitoring of the ecoengineered shoreline. Equally, the effects of ecoengineered seawalls should be compared with multiple unmanipulated seawall sites (Morris et al. 2017).

Monitoring and evaluation programmes can be qualitative, semiquantitative or quantitative and may employ various sampling techniques, including visual surveys, field sensors and destructive sampling (Table 2; Burcharth et al. 2007). Generally, the time required for sampling increases as sampling methods become more quantitative. Which destructive or remote-sensing methods are appropriate will depend on the scale of assessment (e.g. whether changes in alpha, beta or gamma diversity are being evaluated) and the variables being assessed. Core variables for monitoring commonly include the number, diversity and abundance of species (Mayer-Pinto et al. 2018, Strain et al. 2018b), the growth and structure of vegetation or habitat-forming organisms and functional measurements such as filtration rates or primary production (Mayer-Pinto et al. 2018). These variables are typically compared between ecoengineered shorelines, controls and reference areas based on differences in means (Osenberg et al. 1999, Munkittrick et al. 2009, Smith et al. 2017b) and assemblage structures (Anderson 2001). Measures of effect size, such as Cohen's D or log response ratio, can also be helpful (Cohen 1988, Hedges et al. 1999, Coleman et al. 2006, Nakagawa & Cuthill Innes 2007) for quantifying the magnitude of change resulting from the intervention relative to controls and reference sites.

Table 2 Evaluation techniques to measure the ecological and effectiveness of ecoengineered shorelines

Technique	Method	Time scale	Effort and precision	Expectations
Visual	Photographs Videos Online outreach	Hours	Low	Initial documentation of project success or problems to address
Survey data	Structural amount of components (e.g. vegetation, algae, sessile invertebrates, sediments)	Year	Medium	Inform management on initial meeting of project goals
Statistical analysis	Evaluation based on a structured experimental design of target functions	Years	High	Validation of project goals and recommendations on further implementation

To date, much of the experimental research on ecoengineered shorelines has been undertaken at small spatial (i.e. ranging from centimetres to metres) and temporal scales (<12 months) (Bishop et al. 2017, Chapman et al. 2018, Morris et al. 2018b, Strain et al. 2018b). Because of the effects that the shoreline form can have on ecological connectivity (Bishop et al. 2017), and with growing interest by local government agencies, nongovernmental organisations (NGOs) and developers in implementing ecoengineered shorelines at the larger scales required for coastal protection, there is a need to develop appropriate monitoring protocols for addressing ecological changes at these scales. The 12-month monitoring period undertaken by many organisations is arbitrary and probably much too short for demonstrating impacts on community structure or ecosystem functioning. Neither is there sufficient time for natural succession to take its course. For example, in an early study of colonisation of large concrete blocks placed on the outside of a Plymouth Breakwater in the United Kingdom, communities and assemblages took more than 5 years to stabilise to typical small-scale patchiness and low-amplitude fluctuations, with populations of the key grazing limpets taking considerable time (>7 years) to stabilise (Hawkins et al. 1983).

In many cases, the outcomes of ecoengineered shoreline projects at 12 months depend heavily on initial conditions (e.g. created wetland with greater cover will report success in shorter time frames than created wetlands with less cover; Mitsch & Wilson 1996). Research on wetlands suggest that 15–20 years are required to judge the success of ecoengineered shoreline projects (Mitsch & Wilson 1996). The spatial and temporal scales of ecoengineered shoreline monitoring should be linked to the generation time and geographic distribution of the communities being investigated. Developing standardised monitoring protocols and promoting their use across multiple users (e.g. the Shoreline Monitoring Toolbox in Puget Sound, Washington State; http://wsg.washington.edu/toolbox and the Hudson River in New York: Findlay et al. 2018) can help to facilitate collaborations between ecologists and the broader public (Toft et al. 2017) and may help to extend the monitoring time frames of ecoengineered shoreline projects beyond 12 months. But one size is unlikely to fit all cases, and monitoring programmes need to be tailored to the objectives of a project and its environmental and socioeconomic context.

A lack of clear project goals and appropriate monitoring and evaluation of these has hampered assessments of ecological successes and failures. For example, in an analysis of oyster reef restoration projects undertaken in the Chesapeake Bay for shoreline stabilisation or other purposes between 1990 and 2007, only 43% of projects included both the restoration and monitoring required to assess their success or failure (Kennedy et al. 2011). Similarly, of all documented marine coastal restoration projects over a 40-year period, only 61% provided information on survival of the restored organisms (Bayraktarov et al. 2016). Where monitoring has been carried out for ecological enhancements of hard defences, it has typically focussed on the success of interventions in enhancing species richness and/or the abundance of key functional groups relative to controls (see Strain et al. 2018b). Although overall species richness tends to increase with the addition of microhabitats, responses among functional groups can vary. The extent to which increased species richness is due to native versus nonnative species (Morris et al. 2018a, Strain et al. 2018b), as well as effects on ecosystem services (Mayer-Pinto et al. 2018), are rarely addressed, and yet they are important considerations for monitoring, particularly where ecoengineered shorelines are designed to meet multiple ecological objectives.

An additional limitation in measuring the success of ecoengineered shorelines in meeting ecological objectives is that unsuccessful (or only partially successful) projects are rarely documented, making it difficult to assess their frequency (see comments and an example in Firth et al. 2016a). It is likely that the incidence of failure is high, particularly for soft or hybrid ecoengineered shoreline projects in which plantings are conducted without first assessing adequate environmental suitability of the site. For example, between 1989 and 1995, only 1.52% of the 9050 ha of mangroves planted in West Bengal, India, survived (Sanyal 1998). An evaluation of the success of a the $35 million World Bank funded Central Visayas Regional Project in the Philippines indicated that only 18.4% of the 2,927,400 mangroves planted over 492 ha survived (Silliman University 1996). Plantings are especially likely to fail at sites that have not previously supported the planted species, underscoring the importance of collecting data and identifying appropriate ecological objectives prior to the implementation of ecoengineered shorelines.

Measuring engineering outcomes

Ecoengineered shorelines are generally conceived to address one or more engineering objectives, which may or may not be directly related. These include shoreline stabilisation, flood mitigation, energy attenuation and sediment control. Depending on a number of factors, including the physical setting, funding source and socioeconomic considerations, one or more of these engineering objectives may take precedence and must be identified clearly so that appropriate success metrics can be defined. As with ecological objectives, the success of engineering objectives then requires the development and implementation of an appropriate framework for monitoring (Yepsen et al. 2016, Findlay et al. 2018).

Ecoengineered shorelines are generally considered successful from an engineering perspective if they achieve predetermined engineering objectives and they remain intact, such that they continue to achieve that objective over time. The former is typically referred to as the engineering *function* of the project, and the latter the engineering *form*. Engineering function and form are often (but not always) related to each other.

The specific metrics used to evaluate function depend on the engineering objectives of the project (Thayer et al. 2003). Shoreline position is an effective measure of function where shoreline stabilisation is a core engineering objective (Yepsen et al. 2016). Because shoreline position tends to be highly 3-dimensional and extremely dynamic, comparison with control sites is essential (Underwood 1994). It may be possible, however, to use an alternative, less variable measure of shoreline stabilisation, such as the marsh edge or vegetation line (Kreeger et al. 2015). Projects in which flood mitigation is an objective typically utilise flood height and/or extent as a metric for success (Yepsen et al. 2016). Although reduction in flood height compared with surrounding areas is typically easy to measure, reduction in flood extent is often more difficult and may require numerical simulations with the ecoengineered shoreline project compared to without based on past flood events (Yepsen et al. 2016). This can be particularly difficult in built environments, as even small changes in development patterns can drastically alter flood pathways and the extent of flooding.

Projects with energy attenuation as an objective typically use wave height or current velocity attenuation as a measure of success. These parameters are measured in front of (offshore) and behind (inshore) the designed wave attenuation feature and used to calculate transmission coefficients (i.e. the ratio of their measured value inshore versus offshore), which is inversely related to the dissipation of energy (e.g. Garvis 2009, Manis et al. 2015). Even though it is common for the transmission through a traditional engineering structure to increase over time as the structure breaks down, certain types of ecoengineered shorelines, such as those incorporating ecosystem engineers or biogenic taxa, are expected to become more effective over time as the biological growth helps to attenuate the incident energy (Manis et al. 2015). Projects aimed at controlling sediment deposition/erosion typically use areal extent or sediment volume to judge success. Areal extent (the net increase or decrease in land area), which is related to shoreline position, is an essential metric in projects intended to accrete or trap sediment (i.e. with features such as sills, stream barbs, and groynes). While volume change is perhaps a more complete measure of sediment control, it typically requires more effort and expense to collect and therefore it is sometimes forgone.

Metrics related to engineering form provide information about the physical integrity of specific features of ecoengineered shoreline projects. The most appropriate metrics of form depend on the type of ecoengineered shorelines being constructed. *Crest elevation* refers to the elevation of the top of an engineered or natural shoreline feature. Several of the engineering objectives defined here (e.g. wave height attenuation and flood mitigation) are strongly dependent on elevation (Armono & Hall 2003, Allen & Webb 2011, Webb & Allen 2015). Thus, stable crest elevation over time is critical to the success of an ecoengineered shoreline project. In addition to elevation, changes in slope or orientation can be indicative of potential problems. For shore-detached rock structures, changes in slope have a direct impact on the stability of the structure and its ability to dissipate waves. For shore-attached vertical structures, deflections can be indicative of problems related to scour and overtopping. For shore-attached sloping structures, changes in slope directly influence the stability of stone armouring, as well as run-up. *Scour* is a generic term referring to the erosion that occurs adjacent to a hard feature. Scour occurs naturally in front of cliff faces and around rocks, but it also can be caused by built features such as bulkheads and seawalls. It can occur in front of (toe scour), adjacent to (flanking or end effects) or behind (leeside scour) structures.

Another important component of engineering monitoring is inspection of structural performance in terms of integrity of materials or construction units (Burcharth et al. 2006). Structural integrity can be broadly used to describe the physical integrity of an ecoengineered shoreline project, considering the wide range of types of structure this includes. Despite being designed in a manner that facilitates ecological enhancement, often the main objective of an ecoengineered shoreline project is to provide a structural solution. When considering a composite structure, such as a riprap or revetment, structural integrity can be evaluated according to the displacement of individual units from the structure (Burcharth et al. 2006). For solid structures such as a seawall, structural integrity can be about the physical degradation of the overall structure and the constituent components. A *component* is defined as an individual structural member, and the collection of such members such as piles, pile caps, decks and fittings makes up the structure. When dealing with soft and hybrid solutions, such as marshes or marsh sills, some of these components are living, including marsh plantings and items derived from natural materials such as coir logs. In addition to assessing the overall condition of the structure, each individual component should be rated (NYCEDC 2016). Monitoring techniques include visual site inspections (described in further detail later in this review), taking core samples of concrete/rock armour and more standard concrete testing such as chloride or water penetration and compressive strength (Sagoe-Crentsil et al. 2001). In certain cases, more sophisticated measures can be made using probes for corrosion progress and density (Castaneda et al. 2017).

The techniques used to evaluate the metrics discussed here vary depending on the project type, the monitoring budget and the technical abilities of the monitor. Some of the more common tools and techniques are described here; however, the list is far from exhaustive (Yepsen et al. 2016). The simplest approach for measuring distances, areas and volumes is to establish a localised survey grid and to measure distances and elevations with respect to known, fixed points. Although more expensive and requiring more expertise, utilising a survey-grade global positioning system (GPS) can improve both the accuracy (on the order of centimetres, both horizontally and vertically) and the efficiency of data collection compared with simpler approaches. Light detection and ranging (LiDAR), a remote-sensing method, can increase the accuracy and efficiency of surveys even further, but this technology is often out of reach for most ecoengineered shoreline projects due to the expense and expertise required. An emerging approach, which offers many of the advantages of LiDAR at a fraction of the cost, is the utilisation of recreational grade drones and 'structure from motion' techniques. Structure from motion utilises photogrammetric techniques to piece together 3-dimensional information from a series of 2-dimensional photographs (Turner et al. 2016). Some techniques are capable of providing information about slope/inclination, and simple inclinometers can be used to complement low-cost/low-tech techniques.

Simple water-level/wave staffs can be used to estimate water levels and wave heights (Lapann-Johannessen et al. 2015). More advanced techniques include low- and high-frequency pressure gauges, ultrasonic wave/water level gauges and electronic wave/water-level staffs (high frequency is required for wave measurements in shallow water). Current attenuation is probably the most difficult parameter to measure, due to the expense and difficulty involved in obtaining current measurements. On large-scale schemes, satellite imaging can also be used for monitoring the structural integrity of the project (Yepsen et al. 2016).

Ecoengineered shorelines are often designed for a life span of 25, 50 or even 100 years, and thus they require longer periods for success evaluation. Nonetheless, short-term monitoring (1–2 years postdeployment) of key structural performance criteria (such as cracking or corrosion) can provide more immediate measures of success (Thayer et al. 2003). Levels I, II and III describe standard levels of structural inspection and are based on the American Society of Civil Engineers (ASCE) *Waterfront Facilities Inspection and Assessment Manual* (Waterfront Facility Inspection Committee 2015). Level I examination is a visual inspection that does not require any structural components to be cleaned, and so it can be completed most rapidly. Level II examination is focussed on identifying damaged or deteriorated areas that may be hidden by marine organisms or surface materials. Level III examination is more involved and often entails the use of both nondestructive testing (NDT) techniques and partially destructive techniques. This can include extracting material samples (cores) of the concrete or wooden structure for off-site testing or *in situ* surface hardness testing (NYCEDC 2016). For example, the structural performance of bioenhanced concrete pile encasements constructed in Brooklyn Bridge Park to restore the load-bearing capacity of the pier piles, while generating valuable habitat for marine flora and fauna, was monitored 1 and 2 years postdeployment through Level II inspections (Perkol-Finkel & Sella 2015). These findings, as well as the results of the ecological monitoring for the project, were submitted to the New York State Department of Environmental Conservation to validate the structural and ecological performance of the structure. This was required for acceptance of the bioenhanced structure to provide mitigation credits towards the overall project.

As the field of ecoengineering is relatively new and project budgets are often limited after construction, longer-term monitoring data (>5 years postdeployment) are rare. Nonetheless, as risk reduction and engineering performance are key components of many ecoengineered shoreline projects, it is important to conduct such tests and revisit installations several years after deployment. A recent example is the biological and structural monitoring of a bioenhanced breakwater section in Haifa Port, Israel, where Sella and Perkol-Finkel (unpubl. data) revisited the installation 6 years postdeployment. Biological monitoring included on-site visual and photographic surveys, as well as structural monitoring via a combination of visual inspection and core samples for structural integrity of the bioenhanced concrete.

Long-term monitoring plans should also aim to identify and quantify changes in the structural performance due to the increased presence of marine life. As these natural systems develop and their presence increases over time, they will affect the structure's performance and interact with the surrounding environment. For example, marine growth could increase the crest height of a breakwater and further dissipate local wave energy, or alternatively marine growth on a pile encasement could alter flow patterns and change drag forces around the structure (Yacouby-Al et al. 2014).

Measuring socioeconomic outcomes

Few ecoengineered shoreline projects have achieved socioeconomic success, despite the capacity of healthy coastal habitats to provide a plethora of ecosystem services that are valuable to humans (Barbier et al. 2011). For instance, the complex habitat structure provided by vegetated coastal habitats or biogenic reefs creates a nursery habitat for commercially important invertebrate and fish species (Beck et al. 2001, Heck Jr et al. 2003). Shellfish reefs filter nutrients and metals from the water, which can increase water quality (Gifford et al. 2004, 2005, Kellogg et al. 2013, Smyth et al. 2013, Onorevole et al. 2018); similarly, seaweeds can sequester nutrients and heavy metals, as well as carbon (Chung et al. 2011, Henriques et al. 2015, Yang et al. 2015, Krause-Jensen & Duarte 2016). Further, the basis of soft ecoengineering or living shorelines comes from the observation that natural, intact habitats can protect against erosion and flooding (Shepard et al. 2011, Arkema et al. 2013, Temmerman et al. 2013).

There is growing evidence of the efficacy of ecoengineered shorelines in providing and enhancing many of the ecosystem services associated with natural ecosystems (Gittman et al. 2014, 2016, Davis et al. 2015, Onorevole et al. 2018). Nevertheless, their value in providing these services will depend on the identity of the habitat types included in ecoengineered shoreline designs, their density and configuration and the characteristics of the environment in which they are placed (Piehler & Smyth 2011, Smyth et al. 2015). For example, Smyth et al. (2015) found that the relationship between oyster density and denitrification was weakly positive at ambient nitrogen concentrations, but at elevated nitrogen concentrations, it was nonlinear, increasing from low to medium oyster densities but declining at high densities. Overall, it is expected that the ecosystem services provided by an ecoengineered shoreline will increase over time as the habitat becomes established and grows. Even well-established ecoengineered shorelines are unlikely to attain the same level of ecosystem functioning as natural, intact habitats. This is due to trade-offs between engineering, ecological and socioeconomic objectives that result in a design that is suboptimal for ecosystem service provision. Simultaneous monitoring of ecological, engineering and socioeconomic objectives is needed to investigate trade-offs, but it is seldom done.

While there have been attempts to value economically the ecosystem services provided by estuarine and coastal systems (e.g. Costanza et al. 1997, 2014, Barbier et al. 2011), attempts to value the services provided by ecoengineered shorelines holistically have been lacking (Beck et al. 2018, Reguero et al. 2018). Economic valuation involves assigning quantitative values to the goods and services provided by various habitats and is a useful way to compare benefit-cost ratios of management scenarios (Spurgeon 1999). For ecoengineered shorelines, an ultimate economic goal may be that its benefit-cost ratio exceeds that of alternative artificial structures. However, economic valuation can be challenging for ecosystem services because not all of them have a market value. Goods and services with a market value can be evaluated based on revenue increases—for instance, when ecoengineered shorelines increase the production of commercially important seafood species (e.g. Martins et al. 2010), marine-derived pharmaceuticals and raw products, such as firewood. Where there are no markets for services, indirect pricing methods can be used to establish the (revealed) willingness to pay (WTP) or willingness to accept (WTA) compensation for the availability or lack of these services (de Groot et al. 2002).

A variety of indirect methods may be applied to the valuation of ecoengineered shorelines. First, the economic value of shoreline protection may be determined using avoided damages or replacement costs (de Groot et al. 2002, Rao et al. 2015), such as the avoided damage to coastal properties after a storm in areas with an ecoengineered shoreline protecting the coast. This evaluation could be in comparison to a nearby, unprotected shoreline or, alternatively, a nearby artificial structure with a similar exposure (Gittman et al. 2014). The latter contrast enables an analysis of how maintenance and replacement costs following a storm compare between ecoengineered shorelines and artificial structures.

Second, the hedonic pricing model, which works on the premise that the price of a marketed good (e.g. houses) is related to structural, neighbourhood and environmental characteristics, may be applied to ecoengineered shorelines (Landry & Hindsley 2006). Coastal property protected with ecoengineered shorelines may differ in value compared to property with no coastal protection or property protected with hard structures (Landry & Hindsley 2006). House prices are directly related to beach width (Pompe & Rinehart 1994) and other environmental attributes, such as the open/green space provided by wetlands, which can positively influence prices in urban areas (Mahan et al. 2000) and negatively influence them in rural areas (Bin & Polasky 2005). Additionally, house prices are influenced by perceived risk to flooding and erosion (Jin et al. 2015).

A third method for quantifying the value of ecoengineered shorelines is the willingness of a representative sample of individuals to pay for services they provide (de Groot et al. 2002). This can be evaluated in terms of the time and cost that recreational users spend travelling to a site to take advantage of its amenities (Spurgeon 1999). As with the other indirect methods, WTP can be compared between ecoengineered shorelines, shorelines with traditional structures and shorelines without structures. Ideally, WTP may also be compared for a given site before and after ecoengineered shoreline construction.

In addition to such economic valuations, ecoengineered shorelines may be valued in terms of their social benefits and ability to effect policy change. Throughout history, cultures, knowledge systems, religions, heritage values and social interactions have been influenced by the nature of ecosystems (Hassan et al. 2005). Moreover, humans have always shaped the environment to enhance the availability of such services. Ecoengineered shorelines may enable the public to reclaim the water's edge, bringing people back to urban waterfronts (Sairinen & Kumpulainen 2006) for leisure activities and experiences. They can improve shoreline aesthetics, promote public visitation, increase cultural output (Hortig et al. 2001, Reise 2003, Claesson 2011) and be valuable for scientific investigation and environmental education (Krauss et al. 2008, Mitsch et al. 2008). Social impacts can be assessed using a variety of methods, ranging from simple counts of visitation numbers and frequencies and research and education programmes, through to more complex assessments of changing perceptions (Maas & Liket 2011). Questionnaires and participatory mapping of such elements as use values (Brown & Hausner 2017, Strain et al. 2019) are common tools for assessing use and perceptions. Increasingly, technologies such as immersive landscape theatres and virtual reality are also being used to investigate perceptions of aesthetic change (Wang et al. 2013, Miller et al. 2016) and may be suitable for before-and-after assessments.

Policy impacts of ecoengineered shoreline projects take a number of forms (Keck & Sikkink 1998), which makes their measurement challenging. New, highly visible and successful ecoengineered shorelines can provide the first step in influencing policy where ecoengineered shorelines are not currently part of the political agenda. In drawing attention to the utility of ecoengineered shoreline approaches, such projects may persuade governmental or other influential stakeholders to endorse international declarations or conventions, securing procedural change at the domestic and international levels and changes in policy (e.g. changes in legislation or budget allocations). This route could start with the avocation of national or international guidelines for best practices in the construction of shoreline protection schemes.

Procedural changes may include the development of permitting processes or funding schemes that enable the use of and incentivise investment in ecoengineered shorelines for coastal defence. This has the potential to lead to policy changes to support ecoengineered shoreline investments as part of the increased investment in coastal infrastructure. These political effects need to lead to on-the-ground changes in key stakeholders, such as private landholders that are implementing coastal protection works or developers that can declare ecoengineered shorelines as part of the tender (Keck & Sikkink 1998). In the United States, there has been some progress towards investing in living shoreline approaches through policy drivers securing procedural (e.g. green bonds and infrastructure banks used to fund nature-based coastal defence in Massachusetts) and legislative changes (e.g. the Living Shoreline Act in Maryland; Sutton-Grier et al. 2018). There are guides to monitoring and evaluating policy influence (e.g. Reisman et al. 2007, Jones 2011). The method starts with the development of a theory of change (Reisman et al. 2007) that describes a set of activities (e.g. ecoengineered shoreline pilot project), outputs (e.g. design guidelines for ecoengineered shorelines), outcomes (e.g. change to ecological or social system as a result of activity), impacts (i.e. the overall contribution of the outcomes to the goal) and goals (i.e. overall project objective). The outcomes are the target for measuring success, which may be through the impact of research, media uptake, or stakeholder surveys (see Jones 2011).

Adaptive management in response to monitoring outcomes

Assessment of the efficacy of ecoengineered shorelines in meeting objectives, using monitoring and evaluation programmes, is essential to the success of present and future projects. Due to the relative immaturity of the discipline, there is a paucity of data demonstrating how project design and environmental and socioeconomic context influence project success. Such data are critical to informing the design of future ecoengineered shorelines and fine-tuning existing projects to increase their efficacy. Ecoengineering of community structure and ecosystem functioning along urban shorelines is complicated by uncertainties over costs and potential benefits and long-term sustainability (Temmerman et al. 2013), making it difficult for governments to legislate for ecoengineered shorelines. Uncertainty increases over time (Bouma et al. 2014), as natural variability, management objectives and economic constraints can all influence the success of ecoengineering.

Adaptive management is a structured decision-making strategy to govern social-ecological systems that embraces their complexity and uncertainty, providing opportunities for learning and adapting to change (Folke et al. 2005). It rests on the recognition that urban landscapes and seascapes need to be understood and managed as complex, adaptive social-ecological systems and points to the importance of actively managing resilience, here defined as the capacity to persist with functional integrity under changing social and environmental conditions. The feedback between learning and decision-making is a defining feature of adaptive management, with learning contributing to management by helping to inform decision-making and management contributing to learning through interventions that are useful for investigating resource processes and impacts (Williams 2011). Fundamental steps in the process include the articulation of clear objectives (ecological, engineering and socioeconomic), identification of management alternatives, predictions of management consequences, recognition of key uncertainties and quantitative assessment of outcomes.

Interest in and application of adaptive management have grown steadily over the last few decades (Chaffin et al. 2014), and this tool has been recommended as particularly useful in urban ecosystem restoration (Hychka & Druschke 2017) and ecoengineering (Mayer-Pinto et al. 2017). Indeed, the urban setting offers unique opportunities to address environmental issues, while at the same time delivering wide-reaching benefits to the multiple stakeholders and users of marine urban waterfronts. There is also an urgent imperative to create urban infrastructure and environments that are more resilient to climate-related risks. Even though examples of real-scale implementations are limited, scientific knowledge and technologies are being developed to provide adaptive solutions for ecoengineering. Examples include ecosurfaces that can be ecologically retrofitted with further enhancement if monitoring data suggest that the original objectives are not being met (Morris et al. 2018c), optimising the timing and frequency of maintenance interventions to infrastructure by incorporating knowledge of the life histories of species (Airoldi & Bulleri 2011), limiting the social barriers to ecoengineering via better understanding of the social perceptions and expectations through social surveys and interviews (Kienker et al. 2018) and/or providing quantitative data on the cost-effectiveness of various management options (Reguero et al. 2018).

Case studies and scaling up

As is the case for any significant coastal or urban development, scaling up ecoengineered shorelines at any site should make economic sense—so long as *all* costs and benefits are factored into that calculation, including valuing environmental and societal benefits and the costs of traditional development. As with valuation of environmental assets generally, this is still a new field, and the ecoengineered shoreline concept includes a diversity of approaches and technologies. Thus, it is difficult to make any global statements about the cost-benefit ratio for living shorelines across all possible sites and approaches. However, it is instructive to examine projects where the balance between ecology, engineering and socioeconomic factors has played out with varying degrees of success. Three case studies (in Seattle, the East and Gulf coasts of the United States, and Sydney) are described next, and they represent hard, soft and hybrid designs, respectively. Their implementation and evaluation are also outlined. They cover a range of project types, physical settings, engineering objectives and habitats. Although all the projects are multiobjective in nature, the drivers of each project and the metrics used to evaluate them differ.

Case Study 1: Hard ecoengineering: Seattle, Washington

After a large earthquake in 2001, the Elliott Bay Seawall in Seattle, built between 1916 and 1934, was failing and at risk of collapse in the event of another moderate-sized to large earthquake, posing a public safety hazard. Because of the scale of the city's infrastructure, replacement of the wall had an intense, hard engineering focus. However, the need to replace the failing wall with a structure designed to modern earthquake standards also provided the opportunity to ecologically enhance the heavily modified foreshore areas (Cordell et al. 2017). The adjacent estuarine waters are an important migratory corridor for juvenile Chinook salmon (*Oncorhynchus tshawytscha*), which is listed as threatened under the U.S. Endangered Species Act, and the new structure was designed to provide ecological complexity and a well-lit environment conducive to migration, foraging and refuge for salmon and other species, thus incorporating ecological components into traditional poured concrete walls (Munsch et al. 2017a).

Key to the success of this project was social buy-in, as habitat improvement for juvenile salmonids has great cultural, recreational and economic importance in the region. The Seattle waterfront is also a vibrant hub of the city, including many business, tourism, recreation, transportation, shipping and port activities. This social aspect is considerable, especially given the public funding involved on the project and the sense of place that a broad array of user groups depend upon for a connection to the water.

The design and implementation of the Seattle seawall habitat enhancements developed during a decade of studying habitat enhancements for juvenile salmon in Elliott Bay. Information gathered from testing seawall design options with experimental panels that had enhanced texture and slope and overhead light penetrating surfaces (LPSs) allowed for proof of concept at the metre scale that was incorporated into the final seawall design at the kilometre scale (Cordell et al. 2017, Figure 7). Additional components incorporated into the final seawall design came from the success of a project at the nearby Olympic Sculpture Park, which at the 100-m scale created an intertidal bench and a pocket beach (Toft et al. 2013). Data gathered from these smaller-scale projects allowed for scaling-up of ecoengineered shoreline implementation (see the section entitled 'Process of scaling-up successful projects', later in this review, for more discussion).

The final design for Seattle's seawall included: (1) placement of intertidal benches (stone-filled marine mattresses) and creation of an artificial beach, designed to mimic a shallow-water, low-gradient habitat; (2) incorporation of crevices and ledges into the seawall face, with the goal of increasing complexity and enhancing production of invertebrates and algae and (3) addition of LPSs (glass blocks) in the cantilevered sidewalk above the seawall, intended to provide a light corridor to

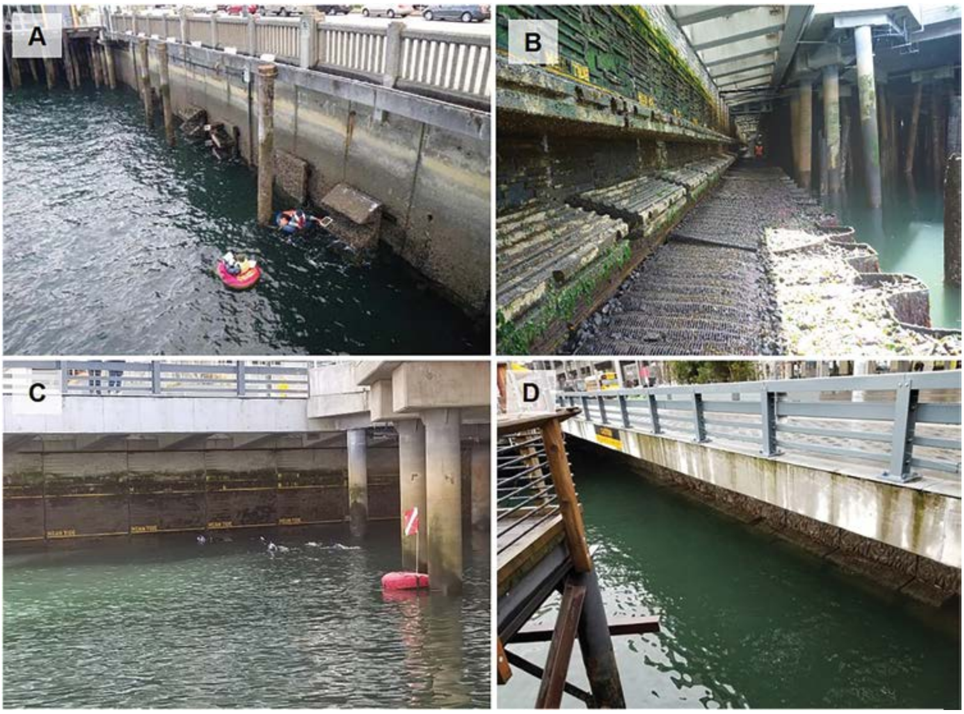

Figure 7 (A) Monitoring algae and invertebrates on experimental habitat test panels on the Seattle seawall before reconstruction; (B) habitat enhancements incorporated into the new seawall, including projections and texturing on the wall and marine mattress benches; (C) snorkelling to survey fish at a mean tide when the bench is inundated; (D) view from an adjacent pier showing a cantilevered sidewalk with light penetrating surfaces consisting of glass blocks.

enhance juvenile salmon outmigration and feeding and to potentially improve productivity under piers (Figure 7).

There is an associated time scale with such a large, hard ecoengineering scheme, highlighting that implementation of ecoengineering projects is not just a matter of spatial scale. Experimental seawall panels were deployed in 2008 and monitored for 4 years, experimental LPSs were deployed in 2013 and monitored for 1 year, the Olympic Sculpture Park was created in 2007 and monitored both before and after enhancement from 2005 to 2011 and the first phase of the Seattle seawall rebuild was completed in 2017, with the second phase yet to be initiated. The experimental stages were needed to ensure the ecoengineered habitats met ecological (i.e. enhanced salmon habitat), engineering (i.e. no compromise to structural stability of seawall) and socioeconomic (i.e. community acceptance) outcomes.

Case Study 2: Soft ecoengineering: living shorelines on the East and Gulf coasts

The 'living shorelines' approach emerged in the 1970s through many examples of bank erosion control and shoreline stabilisation using saltmarsh plantings in Chesapeake Bay (Garbisch & Garbisch 1994). This technique is regarded as successful at providing shoreline protection that is adaptive to sea level rise, in addition to cobenefits such as habitat and food resources for fish and other wildlife or improved water quality (Craft et al. 1999). However, it was recognised that the creation or restoration of saltmarsh alone only provided sufficient coastal protection in low-energy environments. This resulted in the development of the 'hybrid living shoreline' approach, which incorporated varying degrees of hard structure to reduce energy and allow living shorelines to be applied in a wider range of environmental conditions (Garbisch & Garbisch 1994; Figure 8).

Key to the living shorelines technique is that construction must conserve, create or restore natural shoreline functions, in addition to providing coastal defence. In order to retain ecological functioning, strong emphasis is placed on maintaining the connection between terrestrial and marine systems (Bilkovic et al. 2017), a link that is severed by traditional artificial defences, such as bulkheads. In Chesapeake Bay, rock sills are widely used in combination with saltmarsh plantings for additional coastal protection (Figure 8). However, although such a design is regarded as successful for coastal defence, the addition of these hard structures in a soft-bottomed environment results in ecological trade-offs through enhancing filter feeders that colonise the sill at the expense of deposit-feeding infauna, with potential consequences for nutrient cycling (Bilkovic & Mitchell 2013).

There is increasing interest in using oyster-reef living shorelines for coastal protection. In response to the loss of native oyster populations (*Crassostrea virginica*; Beck et al. 2011), restoration techniques, originally applied for fishery enhancement and increasing oyster populations, have been adopted with the primary objective of other services, such as coastal defence and enhancement of water quality. Oysters need a hard substratum for settlement, which led to a diversity of oyster-reef structures being developed. These include loose shell, bagged shell and oyster mats and a number of precast concrete or steel structures (e.g. Oyster Castles®, Ready Reef™, Reefball™, ReefBLK℠, ShoreJAX™, Oysterbreak™ and Wave Attenuating Units®) (Hernandez et al. 2018). Unfortunately there is a lack of scientific or engineering data to support the use of one approach over another. This

Figure 8 Examples of living shorelines projects along the East and Gulf coasts, United States. (A) Restored oyster reefs using oyster mats; (B) bagged shell oyster reef living shorelines with saltmarsh; (C) rock sill with saltmarsh and (D) wave-attenuating units with saltmarsh.

has prevented the development of universal design guidelines appropriate for living shorelines, which has limited their wider application (Narayan et al. 2016, Morris et al. 2018b). Recent studies have tried to gauge public perceptions of living shorelines to help inform management of coastal hazards (Gray et al. 2017). Future research to optimise living shoreline design for coastal protection using the minimum amount of hard material or optimising the placement of hard material for species colonisation (e.g. oysters) will further enhance their ability to provide multiple functions and services.

Case Study 3: Hybrid ecoengineering: Sydney

The emphasis of many foreshore projects is to construct efficiently engineered armouring structures that protect infrastructure, but these frequently have negative impacts on the intertidal environment (Bulleri & Chapman 2010, Ma et al. 2014, Firth et al. 2016b). Projects within the Georges River estuary, including Carss Bush Park Seawall in New South Wales, Australia, have examined foreshore infrastructure design from both scientific and engineering approaches, to not only develop an ecologically responsive foreshore design, but also ensure the successful construction and long-term structural integrity of projects (Figure 9).

The Carss Bush Park project aimed to design and construct infrastructure that allows natural coastal processes, such as tidal ingress, to influence the foreshore. Numerous habitat structures were introduced into what was a highly degraded environment. These included rock pools at varying intertidal levels, longer foreshore slopes and crevices, mudflats for mangrove and benthic organisms, a ‘naturalised’ creek line and endangered saltmarsh benches (Heath 2017, Strain et al. 2018a). The objective of creating such a diverse intertidal zone is to connect the foreshore with previously

Figure 9 Carss Bush Park Foreshore before construction (A and C) and after construction, including saltmarsh (B) and intertidal habitats (D).

developed ecoengineering projects and natural foreshore areas in order to expand the connectivity of estuarine/marine ecosystems within the Georges River (Strain et al. 2018a).

The design and construction of this project have led to improvements in the natural fish and seaweed biodiversity in the artificial rock pools (Heath and Moody 2013, Bugnot et al. 2018, Strain et al. 2018b), while also increasing the social values of the Georges River foreshore (Heath, pers. obs.). Combined with other successful ecoengineering projects along the Georges River, a cumulative improvement in habitat availability has been achieved. The project also incorporates adaptability to future climate change, ensuring its aesthetics and ecological benefits are not affected by sea level rise.

The project design reflected its heavy recreational use and the highly urbanised nature of the location by encouraging the reconnection of the community to the foreshore and Kogarah Bay. This was critical to ensuring the local community's support (Heath, pers. obs.). Landscaping was important in making the ecoengineered shorelines scheme aesthetically pleasing to a broad sector of the community and to promote usability.

The designs also provided environmental benefits, not only with the introduction of intertidal habitats, but with the utilisation of existing seawall sections to create large saltmarsh benches (Heath 2017, Strain et al. 2018a). The use of sections of existing infrastructure in stage 1 reduced the volume of material leaving the site, while also creating the tidal barrier necessary for the saltmarsh to thrive. Critical to the design was ensuring that the saltmarsh was sporadically inundated with saline water during king (spring)–tide events but not affected by all tidal events. By keeping sections of the existing seawall in place, it was not necessary to build new structures or reclaim land from Kogarah Bay as alternative designs required.

While restoration of the Georges River intertidal foreshore to natural conditions is not achievable due to its highly urbanised state (Alyazichi et al. 2015), the foreshore ecoengineering projects outlined previously show how ecological enhancement along a highly urbanised shoreline can be achieved. These ecoengineering projects protect public land from flooding and erosion, improve native biodiversity (Heath & Moody 2013, Bugnot et al. 2018, Strain et al. 2018b) and habitat connectivity (Strain et al. 2018a), with the potential to restore other ecosystem services (e.g. nutrient cycling and carbon sequestration), while developing relationships with the community through educational and recreational engagement (Heath 2017).

Process of scaling up successful projects

To date, most hard and soft ecoengineered shorelines have focussed on rehabilitating or adding a single type of habitat at the site scale (Scyphers et al. 2011, Davis et al. 2015, Strain et al. 2018b). Enhancements to seawalls are typically at the scale of metres or less (Strain et al. 2018b), while living shorelines are typically constructed at scales of less than a hectare (e.g. Scyphers et al. 2011, Davis et al. 2015). In contrast, the scale at which marine ecosystems are degraded by shoreline development is typically in the order of 10–1 million ha (Edwards & Gomez 2007). To benefit large-scale processes and a broad range of species (Bishop et al. 2017), the scale and complexity of ecoengineered shorelines, as for restoration projects more generally, need to be much larger than is commonly considered today (Naveh 1994, Hobbs & Norton 1996, Soulé & Terborgh 1999).

The scaling-up of ecoengineered shorelines may, conceivably, be achieved in several ways. First, beneficial approaches piloted at small scales may simply be applied to larger areas and more sites. For example, the 1.2-km Elliott Bay seawall (in Seattle; see the section entitled 'Case studies and scaling up', earlier in this review) includes textured and sloped surfaces that were initially experimented on at the metre scale (Cordell et al. 2017). Similarly, the approximately 1-km-long New York Living breakwaters project, due to be built following the 'Rebuild by Design' competition (http://www.rebuildbydesign.org/our-work/all-proposals/winning-projects/ny-living-breakwaters), incorporates features such as bioenhanced concrete blocks and tide-pool armouring units that have previously been deployed at demonstration scales (Perkol-Finkel & Sella 2015, Sella & Perkol-Finkel 2015).

It should be noted, however, that because many ecological processes are highly scale dependent (Turner 1989, Wiens 1989, Levin 1992), failure of a particular approach to enhance a desired function at a small scale does not necessarily preclude its efficacy at a larger scale. Hence, interventions need to be designed with the scale of the targeted ecological processes in mind.

There are several reasons why ecoengineered shorelines of larger areas may be expected to yield greater ecological benefits than their smaller counterparts. Ecological theory suggests that the number of species supported by an ecosystem increases with habitat area (see the section entitled 'Links to theoretical and community ecology', earlier in this review; Arrhenius 1921, Gleason 1922). Projects of larger scale are also more likely to incorporate the heterogeneity required to facilitate ecological processes critical to ecosystem function (MacArthur & Wilson 1967, Rosenzweig 1995, Tews et al. 2004). Small areas of habitat are generally considered to be more susceptible to disturbances, as the probability that an entire habitat patch is affected decreases with increasing size (MacArthur & Wilson 1967, Meurant 2012).

Additionally, related to living shoreline projects, wave attenuation increases with distance inside habitat patches, although the magnitude of such protection is the greatest in the first few metres, with the benefits of additional width much diminished (Peterson et al. 2004, Bradley & Houser 2009, Manca et al. 2012). To date, these predictions remain largely untested and the question of how benefits of ecoengineering scale with extent remains a key unknown. Some functions may be expected to scale linearly with areal extent; for example, water filtering capacity may increase linearly with the density or biomass of filter-feeders supported by the ecoengineered shoreline. In contrast, biodiversity, and the ecosystem functions that depend on it, may display a nonlinear pattern of increase to a maximum value, beyond which further increases in area have no effect (Yachi & Loreau 1999, Thébault & Loreau 2006). Monitoring common sets of variables across projects of smaller and larger area would enable future meta-analyses that address relationships between ecoengineered shoreline extent and community structure and function. Furthermore, studies that monitor the benefits of ecoengineered shorelines also require the costs (e.g. facilitation of nonnative species) to determine the optimal area of interventions. Modelling studies that include data on the dispersal capabilities of target and nontarget pest species might assist in identifying the locations at which ecoengineering initiatives may be applied to the greatest benefit (Bishop et al. 2017).

Second, scaling-up may involve increasing the complexity of projects to include multiple habitats. Ecoengineered shoreline projects that include multiple components will increase not only alpha-, but also beta-diversity, and hence ecosystem function (see the section entitled 'Links to theoretical and community ecology', earlier in this review). Many marine species require multiple habitat types in which to complete their life history (Thorpe 1988, Krumme 2009, Sheaves 2009). Hence, building ecoengineered shorelines of a single habitat type may fail to provide the resources needed for species across their life history (Bishop et al. 2017, Morris et al. 2018c). However, whether the inclusion of multiple habitats in ecoengineered shorelines is desirable or not will depend on the identity (i.e. native versus nonnative) and function of the species they support, goals of the intervention (e.g. bolstering the population of a species that depends on multiple habitat types increasing ecosystem function in a severely degraded urban environment, restoring natural habitat configurations) and, potentially, also whether the ecoengineered shoreline is present in an area that historically supported multiple or a single habitat type.

Both the Seattle waterfront (see the section entitled 'Case studies and scaling up') and the New York Living Breakwaters projects exemplify how multiple habitat types may be included in hard-engineering projects. The breakwaters include 'Reef Streets', which add habitat for marine life, as well as intertidal crests, which further increase the complexity and diversity of available habitats. Bioenhancing concrete units are embedded into the breakwaters' fabric, providing targeted enhancement measures. These include multifunction armour units and tide-pools, which by their surface complexity and macro design increase the available surface area to be utilised as habitat and enhance the ecosystem services provided by structure (http://www.rebuildbydesign.org/our-work/all-proposals/winning-projects/ny-living-breakwaters). The GreenShores project in Pensacola, Florida, which created more than 12 ha of oyster reefs, saltmarsh and seagrass habitat along 3.2 km of urban waterfront, is an example of a living shorelines project targeting multiple habitat types (DEP 2012). The sizes of, distances between and identity of habitat patches should be informed by knowledge of the seascape ecology of target species and communities. In addition to the ecological benefits of large-scale projects, there may be significant socioeconomic benefits. As the scale of ecoengineered shoreline projects increases and they are progressively more grounded in science, an economy of scale is expected to emerge. At present, however, the evidence for economies of scale in restoration projects is weak due to their almost universally small scale (Bayraktarov et al. 2016). In some instances, there may even be an inverse economy of scale caused by large-scale projects proceeding without adequate systems knowledge (Turner & Boyer 1997). Additionally, large-scale projects may generate more media and public interest, which may in turn lead to greater opportunities for cofunding and investment.

Several core elements typify projects that have successfully scaled up. These include vision, prior knowledge, technological capacity, financial viability and social licence (Menz et al. 2013). Manning et al. (2006) identify the preemptive constraint of vision as a major impediment to scaling up restoration more generally and suggest stretch goals and backcasting as two potential approaches to overcome this. *Stretch goals* are ambitious, long-term goals that can be used to inspire creativity and innovation (Manning et al. 2006). *Backcasting* involves visualising a desired end point and then retrospectively developing a pathway to that point (Manning et al. 2006). Prior knowledge and technological capacity are commonly built through small-scale pilot studies. For example, the Seattle Waterfront project (see the section entitled 'Case studies and scaling up', earlier in this review), which, among other goals, sought to provide a suitable habitat for juvenile salmon foraging and migration, was designed following years of studying the mechanisms by which urban structures affect juvenile salmon (e.g. Munsch et al. 2014, 2015, 2017a). It built on small-scale demonstration projects, experimenting on the efficacy of proposed interventions at mitigating these impacts (Goff 2010, Toft et al. 2013).

The division of large-scale projects into multiple stages can also assist in ensuring their success. Lessons from earlier stages can be used to inform later stages using an adaptive management approach (see the section entitled 'Adaptive management in response to monitoring outcomes', earlier in this review). Such an approach was used in the Pensacola Green Shores project (DEP 2012).

Worthwhile questions include: What happens if there is failure at the experimental stage? Does it feed back in a different way in the process of scaling-up? Arguably, success and failure are opposite ends of the spectrum of learning that an experiment can provide (Firth et al. 2016a), and both can be used to inform implementation at a larger scale. Instrumental to this line of thought is representing failure in a way that acknowledges the causal mechanisms and generates reasons with an eye to future applications. Social licence is built through engaging stakeholders in all steps of the project, from the planning to the implementation and ongoing monitoring and evaluation phases.

The availability of funding can be a major impediment to large-scale restoration projects (Manning et al. 2006). As previously noted, most research on the ecoengineered shoreline approach has occurred in developed countries (in particular Australia, European countries and the United States). Nevertheless, investment in restoration projects in developing countries can achieve up to 30 times more unit area of habitat than developed countries on a dollar-for-dollar basis, or up to 200 times more unit area when accounting for the local value of the U.S. dollar in developing nations (Bayraktarov et al. 2016). Data on the success of ecoengineered shorelines are urgently needed from developing countries to inform the development of large-scale ecoengineering in these areas.

Concluding comments

In summary, while the protection of all-natural ecosystems remains the ideal conservation strategy, the presence of urban and novel artificial habitats should not be ignored. Given their ubiquity in many countries and increasing prevalence in general, there is a real imperative to compensate for the negative impacts that seawalls and other artificial coastal defences have on shorelines. Examples of ecological engineering efforts around the world have shown that this is possible, but in general, few projects have taken an interdisciplinary approach to setting goals, monitoring and evaluation. This means that there is a paucity of information on the multifunctionality of various ecoengineered shoreline approaches, inhibiting the development of benefits-cost analyses and/or guidelines that could be adopted at national or international scales.

Most of the projects to date have been done on a small scale and in isolation. The adoption of more standardised monitoring techniques that included ecological, engineering and socioeconomic evaluation would enable a more holistic approach to the evaluation and wider implementation of the ecoengineered shoreline approach globally. Proven techniques are urgently needed so that governments, policymakers and international organisations seeking sustainable development goals related to coastlines and oceans have the capacity to develop guidelines, policies and regulations that promote ecoengineered shoreline strategies, where all-natural solutions are not feasible. Greater investment in design, experimentation and development of ecoengineered shorelines could have multiple benefits in improving habitat, biodiversity, water quality and reductions in seaborne waste, resulting in improved access to and appreciation of waterfronts. Doing so would move current options beyond a prime focus on structures with industrial or defensive functions and to a holistic approach with environmental goals with multiple applications at a range of scales.

More examples of successful ecoengineered shorelines are required to demonstrate their potential to provide coastal protection, enhance biodiversity and ecosystem services and increase people's enjoyment. In order to change the way that future coastal defences look and function, ecoengineered shoreline designs generated through interdisciplinary collaborations (e.g. between architects, engineers, ecologists, social scientists and coastal managers) need to be encouraged. Nonetheless, it is important to note that, as in most large-scale engineered intervention, the success of a project at one locality (e.g. Seattle Seawall Project) does not necessarily mean that the same approach will be successful in others (i.e. one size is unlikely to fit all), due to differences in government policies, ecological goals, site-specific environmental conditions, financial support and other socioeconomic factors. As the success of projects from engineering, ecological and socioeconomic standpoints is demonstrated by a sound evidence base, stakeholders will become more invested in ecoengineered shoreline development, initiating the support of government and the development of relevant policy.

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