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Coastal greening of grey infrastructure: an update on the state-of-the-art

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Abstract

In the marine environment, greening of grey infrastructure (GGI) is a rapidly growing field that attempts to encourage native marine life to colonize marine artificial structures to enhance biodiversity, thereby promoting ecosystem functioning and hence service provision. By designing multifunctional sea defences, breakwaters, port complexes and off-shore renewable energy installations, these structures can yield myriad environmental benefits, in particular, addressing UN SDG 14: Life below water. Whilst GGI has shown great promise and there is a growing evidence base, there remain many criticisms and knowledge gaps, and some feel that there is scope for GGI to be abused by developers to facilitate harmful development. Given the surge of research in this field in recent years, we have reviewed the literature to provide an update update on the state-of-the-art of the field in relation to the many criticisms and identify remaining knowledge gaps. Despite the rapid and significant advances made in this field, there is currently a lack of science and practice outside of academic sectors in the developed world, and there is a collective need for schemes that encourage intersectoral and trans-sectoral research, knowledge exchange, and capacity building to optimize GGI in the pursuit of contributing to sustainable development.

Keywords:
Design, Environment, UN SDG 14: Life below water

1. Introduction

The rapid growth of the human population (8 billion in 2022) has driven the intertwined global crises of climate change and biodiversity loss. Much of this growth and associated development has been in the coastal zone, leading to proliferating land-claim and construction of so-called grey infrastructure along the coastline and increasingly offshore into the oceans (Bugnot et al., 2021, Box 1), with many major iconic megadevelopments likely inspired by the construction of the Palms, Dubai (*sensu* the "Palm Effect", Box 2). In many parts of the world, densely populated land-scarce regions (e.g., Penang, Figure 1) may be approaching a form of Malthusian Trap (Malthus, 1798) - where population growth is outpacing land availability and thus expanding built infrastructure into the sea. Such situations may be viewed as valuable for concentrating people in urban areas, but they also present major challenges in terms of resource management and environmental impacts.

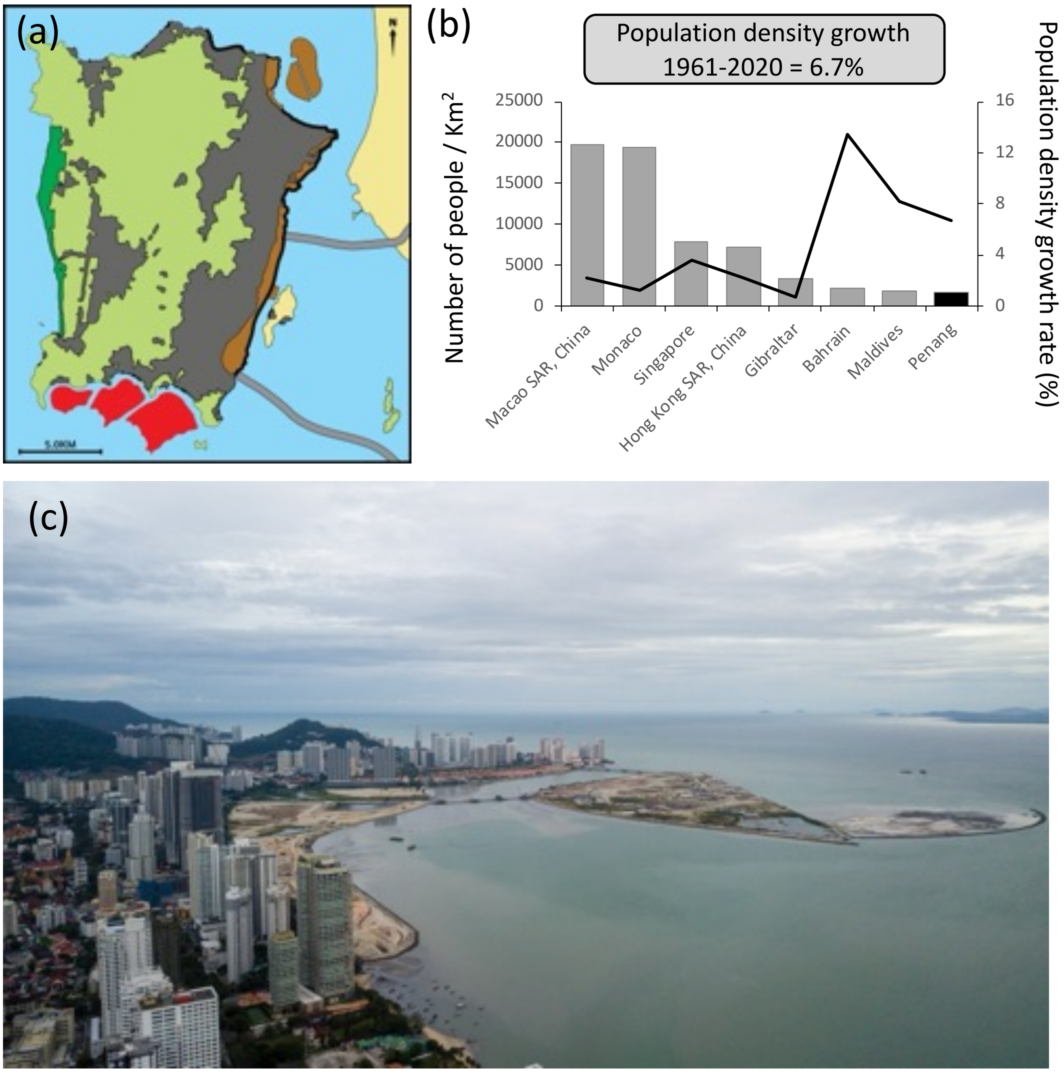

Figure 1. Penang Island: Population growth outpacing land availability? (a) Map showing natural terrestrial habitats (light green = forest; dark green = mangrove), urbanized (grey) and reclaimed land (brown = already reclaimed, red = proposed reclamation). (b) Bars illustrating population density (number of people/km2, 2020 World Bank data) and the lined illustrating population density growth rate (1961–2020) of Penang Island compared to the seven most densely populated countries. (c) Image of tall buildings condensed into a narrow coastal strip between mountains and sea (Photo credit Hong Chern Wern). Figures a and b redrawn from Chee et al. (2017).

The resultant urbanized and degraded environments have been likened to novel ecosystems - hybrids of nature and technology that have been irrevocably deflected from their natural trajectories (Bulleri et al., 2020). The United Nations Decade of Ocean Science and Sustainable Development provides impetus for humans to reverse declines in ocean health and to use science to facilitate sustainable development. Sustainable development is typically managed through the Environmental Impact Assessment and/or the Mitigation Hierarchy processes (Green, 1979). The Mitigation Hierarchy is widely accepted as the current best-practice for achieving sustainable development (CSBI, 2015), wherein practitioners sequentially seek to limit negative environmental impacts and achieve net biodiversity gain (Figure 2). ‘Offsetting’ is intended only as a last resort for developers seeking to compensate for unavoidable damage, after having applied all other steps. With current projections in human population increase, further coastal development is inevitable. Consequently, far greater attention should be given to developing new tools and improving implementation of the Mitigation Hierarchy (Pioch et al., 2017a; Bigard et al. 2020).

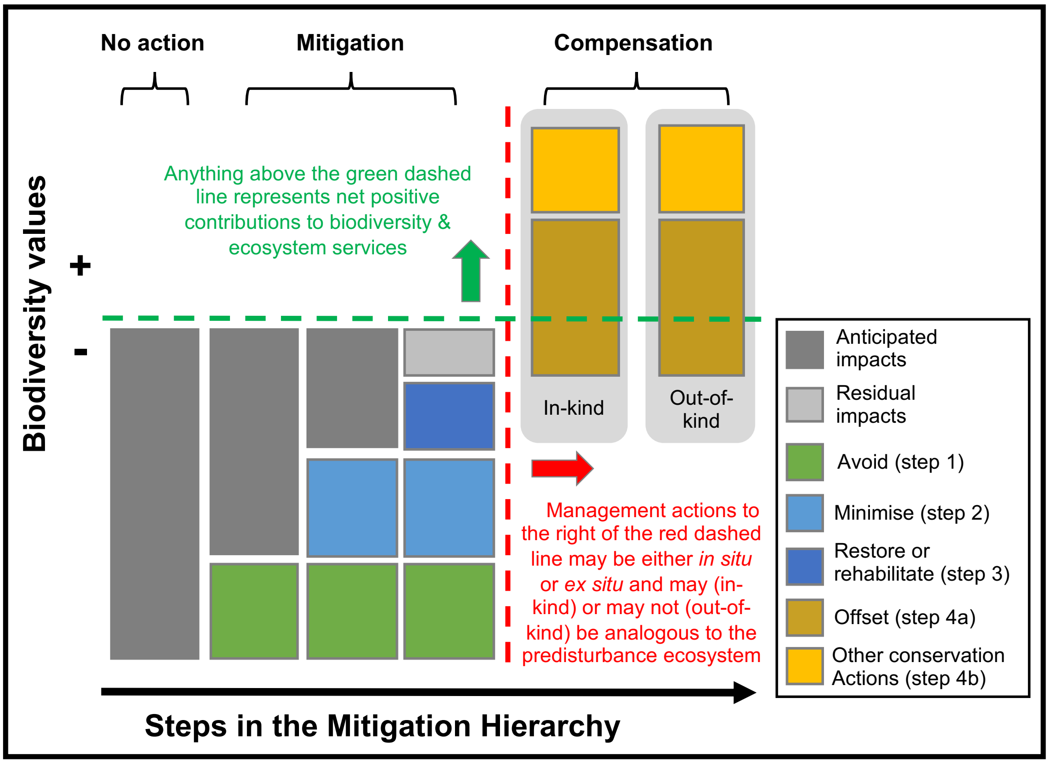

Figure 2. The four sequential steps of the Mitigation Hierarchy: avoid, minimize, restore/rehabilitate and offset/other conservation actions. The goal is to achieve no net loss, or a net gain of biodiversity (not just species richness) (above the green line) compared to the pre-disturbance landscape. Management actions to the right of the red line should be perceived as a last resort and only be implemented when all other steps have been applied to maximum effect. Figure and text adapted from CSBI (2015).

In coastal environments, ecological engineering (eco-engineering) can be used for Greening of Grey Infrastructure (GGI) by incorporating ecological knowledge in designing and engineering multifunctional marine infrastructure (Bergen et al., 2001). The goal is to encourage native marine life to colonize artificial structures to enhance biodiversity, thereby promoting ecosystem functioning and hence service provision (Figure 3). GGI has emerged as a promising tool for achieving biodiversity and environmental benefits (addressing UN SDG 14 – Life below water) with multifunctional infrastructure, such as sea defences, breakwaters, port complexes and off-shore renewable energy installations (see Strain et al., 2018a; O’Shaughnessy et al., 2020a; Airoldi et al., 2021; Evans et al., 2021 for global reviews).

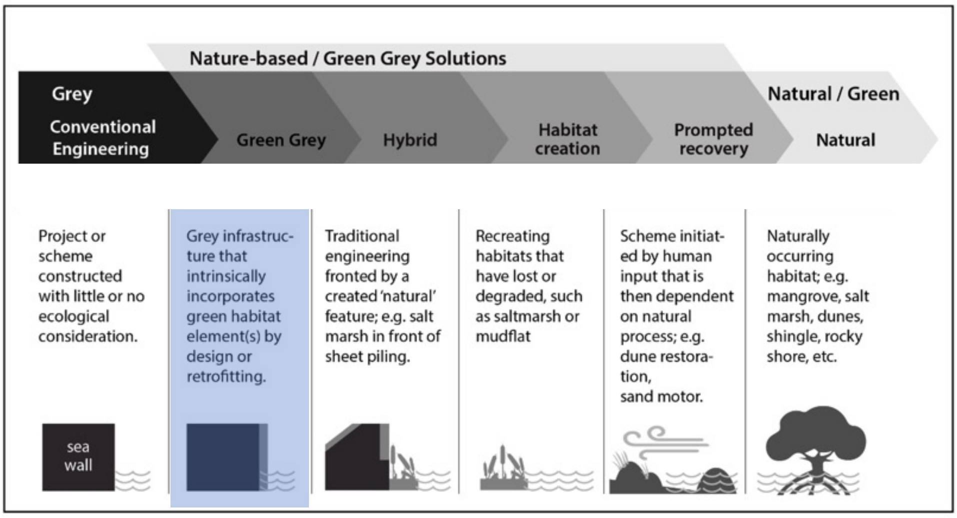

Figure 3. Continuum of marine eco-engineering solutions between conventional grey engineering and natural systems. The blue area shows where Greening of Grey Infrastructure (GGI) sits along this continuum (source: Naylor et al., 2023; adapted from Naylor et al., 2020 and Suedel et al., 2021).

Firth et al. (2020) highlighted a range of limitations and unknowns, noting that GGI interventions could be assigned to one of three categories: Trojan horses, projects that cause environmental damage either through deliberate or misguided intent; Fig leaves, projects that merely cover up or deflect attention from environmental damage caused by the development; and Laurel wreaths, win–win projects with measurable benefits for humans and nature (see Box 3 for examples). Whilst laurel wreaths are more challenging to achieve, this is not to say that all GGI interventions that cannot be classified as "laurel wreaths" are a deliberate greenwashing attempt, as the field is still relatively new and requires more guidance and experimentation.

To prevent GGI from being used for 'greenwashing' to facilitate coastal development, Firth et al. (2020) made several recommendations. Given the upsurge of publications since 2020, in section 2 below we review how scientists and practitioners have addressed these limitations and recommendations. We provide a comprehensive update the state-of-the-art and identify remaining knowledge gaps. In section 3 we address how GGI can be used for greenwashing purposes. Section 4 provides a critical summary of the importance of building the evidence base. Finally, in section 5 we provide some conclusions and make the case for the need for a paradigm shift of current funding strategies and research programmes to encourage the development, implementation, and translation of GGI science at global scales.

2. Update on the state-of-the-art

Firth et al. (2020) made several criticisms and recommendations on how to improve the science and evidence to optimize GGI approaches to and prevent it deliberately or unknowingly facilitating harmful development. Here we expand on and provide a summary of the criticisms and recommendations (Figure 4). To address the major concern that past experiments have been relatively limited in scope, we recommend designing bigger, better experiments. The limited number and scope of response variables typically measured, can be addressed by going beyond simple biodiversity metrics. Furthermore, measuring ecosystem functioning along with socio-economic metrics of service delivery should be incorporated. As most experiments are limited in environmental scope, being conducted in a single location or over short timeframes, revisiting experiments beyond the lifetime of the original project, repeating experiments in the same location where possible and replicating them under different environmental conditions will yield invaluable additional information about how GGI interventions perform under different environmental scenarios. Whilst it may be impossible to fully address the unpredictability of the natural environment, reporting on failures and unintended outcomes of experiments (both positive and negative) will inform better design and save money. Much research to date has been led by the scientific community, and not the practitioner. Science and practice must progress in tandem through collaboration and co-design to ensure that experiments are appropriately scaled-up to 'real world' scenarios, whilst testing their efficacy. Finally, the criticism that there is limited guidance available to practitioners can be addressed through formulating appropriate policies and guidance. Often practice proceeds ahead of science, especially in large-scale projects. Hence developers, planners, architects, and engineers need to be encouraged to engage with scientists to set targets and objectively measure outcomes of designs and interventions intended to lead to environmental mitigation. Such an approach will inform any adjustments or fine tuning required post-construction or commissioning as well as future developments.

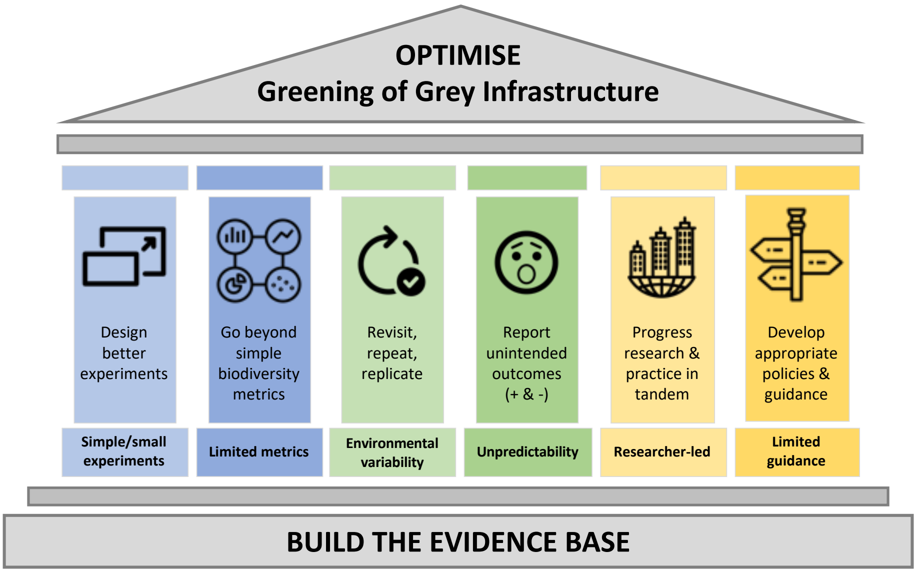

Figure 4. Building the evidence base to optimize Greening of Grey Infrastructure (GGI). The limitations and unknowns are located at the base of the columns. Suggestions and recommendations for a way forward are in the body of the columns (concepts adapted and expanded on from Firth et al. 2020).

In the sub-sections below, we provide a comprehensive update on the state-of-the-art on to address these criticisms and suggestions. See Table 1 for a quick-use reference guide to the most appropriate recent literature that has addressed the various recommendations. A number of comprehensive and large-scale mapping studies have further evidenced statements made by Firth et al (2020) on the prevalence of ambitious large-scale reclamation projects in the Middle-East and Asia, and the vulnerability of African countries to developments stemming from their rapidly rising populations. The major outputs from these are summarized in Boxes 1 & 2.

2.1 Design better experiments

Up until recently, many GGI projects that have primarily been led by the scientific community have been relatively small-scale, simple, or potentially confounded by poor experimental design. For instance, only deploying a small number of units or considering comparisons of a small number of treatments (e.g. microhabitat types or materials) or combining factors that do not enable the researcher to disentangle single effects. Similarly, ignoring differences in surface area between treatments can yield misleading results. Whilst it is well recognized that the incorporation of topographic complexity and microhabitats can provide refugia for organisms from both physical stress and biological pressure, this has rarely been quantified. In the sections below, we focus on the state of understanding on the efficacy of each of these approaches for GGI.

2.1.1 Greater consideration of disentangling the effects of increased surface area and habitat structural complexity

Based on well-known positive effects of habitat complexity and diversity of refugia on biodiversity across scales, many GGI experiments have focussed on incorporating a variety of small-scale microhabitats to increase the settlement, establishment, and survival of different organisms onto artificial structures. Common strategies include retrofitting human-made infrastructure with concrete tiles molded with topographically complex designs (Perkol-Finkel et al., 2018; Loke et al. 2019a; Bishop et al., 2022) or incorporating small-scale complexity into newly built structures (e.g., addition of holes, pits and water-retaining features such as rock pools, Evans et al., 2015; Bender et al., 2020). Early studies enhancing habitat complexity failed to assess the contribution of the increase in surface area (rather than change in complexity) to biodiversity. Loke et al. (2019b) developed a novel system for testing the independent and interactive effects of habitat area and spatial configuration (i.e., fragmentation pattern) on intertidal species richness and revealed an optimal tile density and spatial configuration to maximize biodiversity on tropical seawalls. Whilst a number of recent studies either use small-scale sampling units (Bishop et al., 2022) or compartmentalize the data by microhabitat (e.g., ridges versus crevices on experimental tiles, Strain et al. 2021) in an effort to standardize for surface area, greater attempts to disentangle the effects of increased surface area and addition of topographic complexity is recommended.

2.1.2 Greater consideration for context-specific refuge provision

Arguably, the protection arising from mitigation of environmental stressors and/or predator exclusion by micro-habitat complexity may be more important than the increase in surface area. Outcomes will, however, depend on the type of complexity provided as well as environmental context (Strain et al. 2018a, 2021; Bishop et al. 2022). For example, crevices incorporated into artificial structures can promote oyster recruitment (Strain et al. 2018b), with pits and narrow spaces under overhangs, aiding coral recruitment (Strain et al 2018a; Burt and Bartholomew 2019). For fish, increased structural complexity and provision of diversity of refugia can alter biological interactions (predation and competition), leading to greater fish recruitment and survival rates on more complex artificial structures (Morris et al., 2018a; Burt and Bartholomew, 2019; Bartholomew et al., 2022; Komyakova and Swearer, 2019; Komyakova et al., 2021; Hayes et al., 2022). Understanding the extent to which each of these factors drives biodiversity-habitat complexity relationships is critical to understanding the range of environmental conditions across which habitat complexity enhancement will provide biological benefit (e.g., McArthur et al. 2020, see Living Seawalls, Box 4).

No single GGI intervention is a “one-size-fits-all-solution”. What might be good habitat for one species may be harmful to another (see section below on ecological traps). Similarly, what might be good habitat for one species under one set of environmental conditions may be harmful to it under different conditions. Each species is susceptible to a particular set of predators, competitors and physical stressors that may vary with environmental context. To date, the majority of GGI installations have measured outcomes for biodiversity generally. Few studies have focused on particular species, traits or role in ecosystem functioning. For species that live in the intertidal zone, GGI that provides protection from physical stressors such as temperature fluctuations and solar radiation may be more important on the upper shore, whilst GGI interventions that provide a refuge from predation or grazing pressure (e.g., Martins et al., 2010) may be more important on the low shore and in the subtidal zone (Bishop et al., 2022). Much can be learnt by considering how individual species or functional groups respond to different habitat types under different environmental conditions (Strain et al., 2021; Aguilera et al., 2023; see also section on revisit, repeat, replicate). Whilst some progress has been made in the consideration of environmental stressors and biotic interactions, this remains a major knowledge gap. Successfully identifying species and functional groups that have important structuring roles (e.g., predators, competitive dominants, habitat-formers) in a local context is of critical importance. For this to be successful, designers of GGI interventions need to work closely with ecologists who have an in-depth understanding of the local ecosystem.

2.1.3 Wider consideration of material and substrate types

The growing body of interdisciplinary research involving ecologists and materials scientists is revealing limited differences in the biological colonisation of materials, when cement content, cement-replacements (pulverised fly ash, PFA; ground granulated blast furnace slag, GGBS, fiber concrete), or aggregates and admixtures on biodiversity were examined (Kress et al., 2002; McManus et al., 2018; Becker et al., 2020; Hsiung et al., 2020; Vivier et al., 2021a; Bone et al., 2022a; Hayek et al., 2022; Lapinski et al., 2022). Whilst an early study by Perkol-Finkel et al. (2014) reported greater colonisation on lower pH concretes (pH 9-10.5) than ‘standard’ values (pH 12.5-13.5), this study was confounded as it was impossible to disentangle the separate effects of complexity and material. Recent experiments in temperate and tropical regions (Hsiung et al., 2020; Lapinski et al., 2022) have found no effect of concrete pH reduction on colonization.

Natural rock, metal, oysters, and wood are also extensively used in marine infrastructure. Experiments suggest limited differences in the communities settling and establishing on these materials (reviewed by Dodds et al., 2022 and on coastal concrete by Bone et al., 2022a). Hartanto et al. (2022) for example, reported no tropical intertidal faunal differences and minor algal differences in colonisation of granite, limestone, sandstones and concrete. Little is known about how materials affect microbial community composition. Natanzi et al. (2021) reported differences in the relative abundances of cyanobacteria, diatoms associated with different concrete mixtures after one month. Conversely, after a similar deployment time (31 days), Summers et al. (2022) found no differences in microbial diversity among the same stone types tested in Hartanto et al. (2022). Whilst it appears that surface properties may influence microbial communities early on, it appears that communities converge later in the ecological succession process. Other factors, such as site or surface aspect (north-south directionality) and orientation of material (e.g., horizontal/vertical), may have greater influence on community composition (Firth et al. 2015; Amstutz et al., 2021, 2023), with effects potentially even larger than those from surface complexity (Grasselli and Airoldi, 2021). This evolving picture can help industry focus resources on other aspects of material choice, including carbon footprint (Dennis et al. 2018; Dauvin et al. 2022) and chemical pollution of concrete (Kress et al., 2002; McManus et al., 2018).

2.1.4. Manipulation of mobile invertebrates

The transplanting of mobile invertebrates has received little attention, but Morris et al. (2018a) found that most individuals migrated out of experimental units. Species-specific behaviour and the use of enclosures should be considered for transplantation studies. Conversely, Firth et al. (2023a) suggest that the small-scale removal of limpets from carefully managed patches may yield biodiversity benefits with implications for ecosystem functioning and service provision. Importantly, this suggestion is relevant in a northwest European context, where limpets have a key structuring role (Coleman et al., 2006; Firth et al., 2021a). Removal of limpets in other systems may have little to no benefit and may be detrimental. Any manipulation of species requires consultation with local ecological experts.

2.2 Go beyond simple biodiversity metrics

The benefits of GGI interventions for biodiversity are typically assessed using metrics comparing differences in mean numbers of species (i.e., alpha diversity, Evans et al., 2021). Much can be learnt from consideration of a wider variety of diversity metrics such as beta diversity, which gives a measure of variation in community structure among a set of sample units and hence the importance of variety of micro-habitats on a more landscape scale (Firth et al., 2016b; O’Shaughnessy et al., 2023). More worrisome is the lack of attention to species identity, since it implies that attracting a pest, non-native or ephemeral species is potentially given the same positive weight of attracting species that are rare, on the brink of extinction and/or recognized as in need of protection. Also, an unbiased evaluation of the success of a given eco-engineering strategy requires the set of species one wishes to attract to be clearly identified a priori.

2.2.1 Greater consideration for ecosystem functioning

Limited research effort has focussed on the influence of GGI interventions on ecological functionality, including fitness and reproduction, trophic structure (but see Espinosa et al., 2020; Sedano et al., 2020a,b; Raoux et al., 2022) or ecological connectivity. GGI interventions may for example, serve as hubs for sources of propagules (Reddy, 2022), or facilitate novel interactions among species (Klein et al., 2011) through a failure to support viable populations of key intertidal grazers (Moreira et al., 2006). A key opportunity of GGI is the capacity to deliver multi-faceted ecosystem functioning that can, in turn, support ecosystem services and mitigate stressors common in urban coastal environments (Pioch and Souche, 2021). For example, GGI can assist in reducing wave overtopping on seawalls, increasing their capacity for coastal protection (O'Sullivan et al., 2020). GGI can also enhance the abundance of filter feeders, such as oysters, mussels and sponges, or microbes involved in nutrient cycling, potentially improving water quality (Rouse et al., 2020; Vozzo et al., 2021a; Bulleri et al., 2022; Dodds, 2022). Conversely, added topographic complexity may have positive benefits on species, but have little or no effect on productivity (Mayer-Pinto et al., 2023); and results can be location specific (Mayer-Pinto et al., in review). Artificial structures impact surrounding habitats via changes in the characteristics and biodiversity of proximal sediments (Hanley et al., 2014; Heery et al., 2018; Martinez et al., 2022), transport of wrack detritus (Critchley et al., 2021), litter accumulation (Aguilera et al., 2016, 2023; Aguilera 2018), and novel use of structures by terrestrial (pest) predators (Aguilera et al., 2023a). Initial work suggests that GGI can ameliorate some of these impacts, by retaining wrack (Strain et al., 2018b), but the influence of GGI on sediment properties (but see section 2.4.3 below), biotic assemblages and predator-prey interactions remains largely unknown.

2.2.2 Greater consideration for ecosystem services

GGI interventions may furthermore provide social and economic benefits such as aesthetic landscape appreciation (Pioch et al. 2011) and fisheries enhancement (Morris et al., 2018a; Burt and Bartholomew, 2019), especially when incorporating indigenous knowledge (Porri et al., 2023). A large-scale example is increased habitat use and feeding by juvenile salmon on large-scale GGI in Seattle (Sawyer et al. 2020; Accola et al.;2022a,b). Generally, however, the economic valuation of ecosystem services associated with GGI is in its infancy (e.g. Mehvar et al., 2018), but is nonetheless essential to support or challenge economic justifications by enabling cost-benefit analysis (Fairchild et al., 2022). Economic values include societal benefits via greater knowledge gained of GGI (Strain et al. 2019), a perceived increase in naturalness and biodiversity, as well as an association with healthy environments (Fairchild et al., 2021; Salaün et al., 2022). Perspectives may, however, vary among stakeholders and regions; hence further studies are required for a more comprehensive understanding of perceived benefits and potential social and ecological conflicts in implementing GGI (Morris et al., 2016; Pearson et al., 2016; Kienker et al., 2018; Salaün et al., 2022; Aguilera et al., 2023b). It is important to acknowledge that trade-offs between positive ecological outcomes should be considered. In an experiment where both physical and biological complexity were added to seawalls, trade-offs between species richness and functional outcomes were observed (Mayer-Pinto et al., 2023). Moreover, GGI may negatively affect surrounding natural habitats. If, for example, GGI promotes novel communities with high filtration capacity as observed on artificial structures (Layman et al. 2014), it can potentially impact marine food webs beyond the footprint of the structure (Malerba et al., 2019; Raoux et al., 2022); conversely societal benefits may be delivered by increasing water quality, especially in enclosed urban water bodies (Wilkinson et al. 1996). Further research is therefore needed in this area (Riascos et al., 2020), including differing viewpoints of different constituencies of stakeholders.

2.3 Revisit, repeat, replicate experiments

A growing number of studies are testing the efficacy of GGI interventions across local gradients and biogeographic regions (e.g., Strain et al., 2021; Clifton et al., 2022; Mayer-Pinto et al., in review). Spatially replicated experiments have revealed that impacts of many (but not all) GGI interventions are highly context dependent, with results ranging from positive to neutral to negative, across spatial scales of centimeters to hundreds of kilometers (Strain et al., 2021; Chee et al., 2021; O'Shaughnessy et al., 2021; Clifton et al., 2022). Whilst Strain et al. (2021) is exemplary in its spatial extent (28 sites, in 14 cities, spanning 5 continents globally), this was a short-term experiment that relied on each partner independently 'buying in' and having the financial and human resources to contribute within a particular timeframe. Consequently, the temporal extent was limited to just 12 months. Nonetheless, this model of global replication is the 'gold standard' and should be aspired to when and if global funding opportunities enable such projects. Replication of previous experiments at new locations also shows that results cannot be generalized; in contrast to experiments conducted by Perkol-Finkel and Sella (2014) in Israel, experiments by Hsiung et al. (2020) in the UK and Singapore showed concrete pH to be unimportant in determining biodiversity. Similarly, whereas artificial rockpools have strong positive effects on biodiversity in many temperate settings (Browne and Chapman, 2014; Evans et al., 2015, Ostalé-Valriberas et al., 2018; Hall et al., 2019), in some tropical settings, extreme high temperatures, precipitation and sedimentation may limit their benefits (Firth and Williams 2009; Waltham and Sheaves, 2018; 2021, but see Chee et al., 2020). In a study spanning five locations across three continents, Mayer-Pinto et al. (in review) found conflicting results for GGI interventions on productivity and respiration metrics. Many GGI interventions (especially those that need to be affixed to infrastructure, such as tiles, panels and precast units such as vertipools and pots) are susceptible to damage from wave action (Browne and Chapman, 2011), interference by sediment inundation (see section 2.4.3 below), and even total burial by sand, particularly of the intervention is associated with shore perpendicular groynes (e.g., the BIOBLOCK, Firth et al. 2020). Conducting experiments along wave exposure gradients will yield invaluable information about limiting conditions and engineering constraints.

For eco-engineering interventions based on habitat complexity enhancement, key correlates of spatial variation include latitude, tidal height, size of the local species pool, and locally dominant stressors (Strain et al., 2021; O'Shaughnessy et al., 2021; Clifton et al., 2022). These factors are likely to reflect the varying importance of habitat complexity in mitigating key environmental stressors (e.g., extreme temperatures, desiccation) and intensity of biological interactions (predation, competition, facilitation) in different settings (Strain et al., 2018a). They will also reflect environmental variability in the identity, diversity and supply of colonists on which habitat complexity can act (Clifton et al., 2022). The type of habitat complexity provided (e.g., holes, crevices, water-retaining features) also influences eco-engineering outcomes (Strain et al., 2018a; Bishop et al., 2022). Thus, identification of natural complexity patterns on different rock types and how they vary across latitudes, wave exposure gradients and tidal ranges are relevant in this context to inform eco-engineering approaches (e.g., latitudinal gradients in microhabitat availability; Bracewell et al., 2018; Aguilera et al., 2022). Biomimicry-inspired designs, by learning from nature can inform both the ecological functioning and aesthetics of GGI (Pioch and Souche, 2021); habitat-building species may provide the most profitable avenues to explore (Byers 2022).

Despite increasing spatial replication (see Box 4), most GGI studies remain limited to less than one year (reviewed by Strain et al., 2018a, Dodds et al., 2022; but see Wilkinson et al., 1996, Bender et al., 2020, Chee et al., 2020, Bishop et al., 2022 for exceptions). The importance of GGI interventions for biodiversity can change over time. Both Martins et al. (2016) and Bender et al. (2020) revealed similar positive results on revisiting GGI installations after six and twelve years respectively (see Martins et al., 2010 and Langhamer and Wilhelmsson, 2009 for original studies). Importantly, both studies specifically targeted enhancing species of commercial interest. Less is known about the long-term influence of GGI interventions on biodiversity more broadly. Where rates of succession are slower (e.g., in temperate regions), early colonization processes remain over-represented in the GGI literature. Meta-analyses report weak (non-significant) patterns of diminishing substrate property and habitat complexity effects through time, and more multi-year studies are needed to adequately explore this (Strain et al., 2018a; Dodds et al., 2022). Indeed, in addition to the confounding impacts of competition noted above, Bishop et al. (2022) reported diminishing effects of physical habitat complexity on species richness after one year, because habitat-forming taxa themselves became the key determinant of habitat complexity. Early and mid-successional opportunistic species may also inhibit later colonizers, unless their dominance is broken by physical or biological disturbance (Sousa 1979). Besides evaluating the benefits of eco-engineering at ecologically meaningful scales, long time series are critical in assessing GGI performance under rapidly changing environmental conditions (e.g., warming associated with climate change, Sun et al., 2022; Waltham et al., in review).

2.4 Report on unintended outcomes

All experimental trials and installations have the potential to become damaged and/or lost due to a range of natural and anthropogenic factors. For instance, tiles or units that are affixed to seawalls may suffer damage from wave action and storms (Browne and Chapman, 2011), or vandalism from members of the public (de Moraes et al., 2022). It is important that researchers and practitioners alike report on such incidents, particularly in relation to factors that can be controlled, such as choosing a location that is less likely to be damaged by wave action. Arguably, it is more important to know when schemes fail than when they succeed, as greater knowledge is gained from failure than from success. In the sections below, we review some recent advances in some of the unintended outcomes that were identified by Firth et al. (2020).

2.4.1 Ecological traps and happy accidents

‘Ecological traps’ occur when the links between habitat quality and habitat selection are decoupled, resulting in negative fitness outcomes (Battin, 2000; Komyakova and Swearer, 2019; Swearer et al., 2021). Depending on the scale of impact and the species in question, ecological traps may cause local and regional extinctions (Hale et al., 2015). The question of ecological trap formation due to GGI installations has received little attention. Recent research on artificial reefs (Komyakova et al., 2021) and fish farms (Barrett et al., 2018, 2019) provide some evidence for concerns. Habitat selection is generally based on suitable habitat detection using a range of cues (Kingsford et al., 2002). Increased temperatures and pollution (e.g., chemical, light, and noise) can impede the ability of larvae to differentiate positive and negative selection cues (Doney et al., 2009; Nilsson et al., 2012) leading to poor habitat selection decisions with negative fitness consequences (Fobert et al., 2019; Komyakova et al., 2022;

Marangoni et al., 2022). Consequently, GGI installations deployed in polluted environments (e.g., ports, marina, and harbors) may enhance the likelihood of ecological trap formation (Komyakova et al., 2022). Additionally, if GGI is applied to marine infrastructure where recreational fishing is popular, higher mortality rates for certain species may result (Swearer et al., 2021); perpetuating the attraction over production argument (Pickering and Whitmarsh, 1997). Further research is urgently needed to understand the implications of increased attractiveness of GGI installations and potential fitness consequences. Importantly, artificial habitats can act as ecological traps for some species and population sources for others, and hence a multi-species approach is needed (Komyakova et al., 2021).

Some coastal infrastructure can have unintended positive outcomes or 'happy accidents' (Rosenzweig, 2003) for species of conservation concern. In South African estuaries, marinas are constructed with gabions (wire cages filled with rocks) and lined with reno mattresses (flattened gabion boxes used to line canals for erosion control). Originally chosen for aesthetic purposes, the gabions and reno baskets have serendipitously provided habitat to the endangered Knysna seahorse (*Hippocampus capensis*) (Claassens, 2016; Claassens and Hodgson, 2018) and are even used by *H. capensis* (and other species) in preference to natural eelgrass habitat (Claassens et al., 2018). Similarly, seawalls in the port of Ceuta, North Africa appear to be important refuges for the endangered limpet, *Patella ferruginea* from human harvesting (Rivera-Ingraham et al., 2011). Where coastal infrastructure supports species of conservation concern (see Firth et al., 2016a and Ostalé-Valriberas et al., 2022 for reviews), attempts should be made to deploy GGI interventions to boost populations (e.g., Martins et al., 2010; Langhamer & Wilhelmsson 2009) and to manage such sites as part of wider networks (e.g., artificial marine micro reserve networks, García-Gómez et al., 2011, 2015; Ostalé-Valriberas et al., 2022).

2.4.2 Invasive non-native species

Convention suggests that GGI installations can promote the diversity and abundance of native species, with the view to offsetting invasion success of non-native species (NNS, Stachowicz et al., 1999; Arenas et al., 2006). There is no reason, however, why an increase in habitat complexity (either physical or biological via seeding/transplanting of habitat-formers) should not also favour NNS (Gauff et al., 2023). Evidence of NNS responses to GGI approaches is growing (e.g., Peters et al., 2017; Perkol-Finkel et al., 2018; O'Shaughnessy et al., 2020a) with variable outcomes across geographic locations, tidal height, types of interventions and functional groups or species. NNS responses to GGI interventions are inevitably species-specific depending on the environmental tolerances and traits expressed by individual species. Rather than looking at the diversity of NNS across experimental units, researchers could consider diversity/abundance of native species as covariates in analyses, and how patterns of colonization by natives interacts with experimental manipulation in NNS response.

Early evidence that GGI interventions could enhance native biodiversity over NNS came from "bio-enhanced panels" (Perkol-Finkel et al., 2018), although, as both surface complexity and concrete composition were manipulated in the "bio-enhanced panels" individual effects cannot be distinguished. More recent GGI studies replicated in different geographical locations (World Harbour Project) isolated complexity from other factors with contrasting results. For example, in Plymouth, UK, O'Shaughnessy et al. (2021), reported reduced NNS abundance on complex compared to flat tiles deployed subtidally; a similar result emerged for intertidal treatments in Sydney, Australia (Vozzo et al., 2021a). By contrast, no treatment-specific differences emerged for NNS in either sub- or intertidal experiments in East London, South Africa (Mafanya, 2020) despite the presence of NNS in the local area (Peters et al., 2017). Using the same panels in Sydney, Australia, Schaefer et al. (2023) found that the more complex tiles supported greater abundance of invasive ascidians, particularly when manufactured with oyster shells compared to controls or those manufactured with vermiculite. Similarly, Gauff et al. (2023) working in Toulon, France, found that complex habitat, engineered to protect juvenile fish from predation, increased NNS numbers. Less information on GGI and NNS is available from the tropics, but reports from several major shipping ports have found few to no NNS (Waltham and Sheaves, 2021; Tan et al., 2018; Wells, 2018; Wells et al., 2019; Wells and Bieler 2020).

The ‘priming’ of grey infrastructure by the addition or ‘seeding’ of native habitat-forming species to reduce NNS recruitment has also been investigated but has again generated mixed results. O'Shaughnessy et al. (2021) found no difference in numbers of NNS across seeding treatments, while Vozzo et al. (2021a) reported how GGI seeding with a native oyster increased the abundance of a non-native isopod. Morris et al (2018b) however found that the seeding of water-retaining rock pools with mobile invertebrates limited NNS establishment.

Despite limited evidence that GGI promotes NNS diversity or abundance, practitioners and government agencies remain concerned that GGI may contribute to the introduction and spread of NNS. Whilst it might be impossible to predict or prevent colonization of NNS, to minimize risk of colonization, it is necessary for GGI to be designed to inhibit colonization of pest species of local concern (see Dafforn, 2017 for review). This requires knowledge of the local species pool and potential impacts to native biodiversity. Furthermore, knowledge of NNS hotspots linked to shipping patterns may be informative (Tidbury et al., 2016; O’Shaughnessy et al., 2020b).

2.4.3 Effects of sedimentation

GGI interventions can be prone to sedimentation, limiting their capacity to host typical hard-substrate biota (Firth et al., 2016b; Hall et al., 2019; Waltham and Sheaves, 2018; Bone et al., 2022b). Although considered a potential management issue, due to the perceived costs and need for responsibility associated with sediment removal (Waltham and Sheaves, 2018), sediment collected in artificial rockpools in the UK contained infauna typical of surrounding estuary mudflats (Bone et al., 2022b). Consequently, it is important to consider retained sediment as potential habitat and to sample accordingly, even if the retained sediment is perceived to be less ‘interesting’ than rockpool habitats by the public or practitioners (Bone, pers. comm.). Changes in attitude towards retained sediment begins with the scientists themselves (often with backgrounds in rocky shore ecology) viewing sedimentation as ‘potential’ instead of ‘problem’. Nonetheless, undesirable sedimentation could be avoided by greater understanding of the local sediment supply and the depositional environment prior to GGI installation. The inherent dynamism of coastal environments means that features previously free of sedimentation may suddenly become inundated, and vice versa. Given that infrastructure often causes loss of sedimentary habitats (Heery et al., 2017), much greater attention should be given to GGI interventions that encourage the accumulation of sediments, but on sufficiently large scales that they support fully functional habitats.

2.5 Research and practice should progress in tandem – scaling up

The GGI concept has been driven since the early 2000s by an ecological research perspective showing that shoreline armouring has significant impacts on biodiversity and functioning (Chapman, 2003; Airoldi et al., 2005; Mayer-Pinto et al., 2018). Integrating GGI into everyday practice requires interdisciplinary collaboration and co-design - combining the necessary engineering and ecological expertise to ensure GGI success and incorporation into technical standards (Pioch et al. 2018). Unfortunately, logistical and financial constraints have limited GGI experiments to small-scale (i.e., few meters of the complete infrastructure) projects retrofitted onto existing structures, with few examples globally of research uptake into industry-led projects (but see Box 4 on Living Seawalls).

The Seattle seawall project showcases the integration of research into practice. Small-scale trials of complex concrete panels and a bench installation at the seawall base to test GGI approaches provided shallow-water habitat and enhanced prey for migrating juvenile salmon (Toft et al., 2013; Cordell et al., 2017; Sawyer et al., 2020). These techniques were later upscaled to nearly 1-km of upgraded seawall, with the addition of skylights into the boardwalk to increase light penetration to the migration corridor. It was estimated that the incorporation of GGI added 2% cost to the US $410 million build (Sawyer et al., 2020; Accola et al., 2022a), but the costs of long-term maintenance is unknown. This project was co-designed by ecologists and engineers from the onset, with a clear objective to improve fish habitat while providing the coastal protection required, and human access to and appreciation of the marine environment. As such it provides an aspirational benchmark and standard, not only for the application of GGI initiatives, but multifunctional infrastructure globally.

2.6 Develop appropriate policies and guidance

Key to avoiding greenwashing is the co-creation of guidance, case studies and policy instruments with practitioners (Naylor et al. 2012) in several domains (e.g., flood risk, green infrastructure) where eco-engineering interventions and implementation measures are placed on a spectrum (Figure 2, Pioch et al., 2011; Perkol-Finkel and Sella, 2014; Taljaard et al., 2019; Naylor et al., 2023). Levers that could support increased uptake of eco-engineering are documented (e.g., Naylor et al., 2012, Pioch et al., 2018; Evans et al., 2019; Mayer-Pinto et al., 2017; Claassens et al., 2022) and policy gaps identified (e.g., Victoria's (Australia) Coastal Strategy, 2020 and South Africa's Coastal Management Act, 2014, RSA, 2014; Global Inventory of Biodiversity Offsetting Policies, GIBOP, 2019). To avoid greenwashing, specific guidance and co-created case studies (e.g., International Guidelines, Suedel et al. 2021; NOAA's Habitat Blueprint), strategic plans (e.g., National Marine Science, 2014), state of science (Australian Government, 2021), regulatory requirements (e.g., Hydraulic Project Approval) and policy instruments (e.g., Welsh Government, 2021) are crucial. Flood management is the policy domain where the greatest efforts have been undertaken to limit greenwashing and encourage uptake of green-grey eco-engineering (e.g., The Washington State Hydraulic Project Approval, USA). The United Kingdom is a leader in this space, with eco-engineering (and GGI) guidance (Naylor et al., 2012, 2017; CIRIA, in prep) and co-produced case studies (Estuary Edges, 2008; Naylor et al. 2017) leading to statutory flood policies (Welsh Government, 2019, 2021), explicitly stating that GGI should only be implemented where other nature-based options are not suitable. Similar inventories of case studies and guidance exist for Singapore (Lai et al., 2022) and Malaysia (Chee et al., 2021b). In 2023, the French ministry will publish a guideline to favour GGI and eco-engineering practices in future French ports (DGITM, 2022). Other policy domains (e.g., conservation, infrastructure, marine) would benefit from explicitly mentioning green-grey eco-engineering.

3. Greening of grey infrastructure and greenwashing

Central to Firth et al. (2020) was a concern that GGI could be misused for greenwashing as consultants, developers, and local authorities implement GGI to expedite, facilitate, and reduce costs of regulatory processes. Not only does this remain the case, but reports from municipal engineers (Firth, pers. comm.) also indicate how by strategically incorporating GGI onto marine infrastructure on existing reclaimed land, public opinion can be swayed in support of proposals for future large-scale land reclamation. This we argue, is a clear example of GGI being used as a Trojan horse for environmentally harmful development. In response to greater take up of GGI applications on coastal infrastructure, private companies that design environmental products have emerged globally. Although vital to successful GGI implementation and development of local economies, self-evidently, companies that design and produce environmental products such as habitat enhancement units, novel concretes and artificial reef products stand to gain financially from greater uptake of GGI schemes. We strongly recommend objective and independent evaluation of proposed GGI installations and for a critical assessment of development monitoring and assessments funded or initiated by those with a conflict of interest. Furthermore, any data generated should be publicly available to ensure transparency.

4. Building the evidence base

It is imperative for environmental management decisions and actions, such as GGI, to be evidence-based (Downey et al., 2022; Lemasson et al., 2023; Sheaves et al., 2021), to maximize beneficial outcomes and minimize waste of time and resources, mainly from public funds (Sutherland and Wordley, 2017). In the nascent field of GGI, we must remain particularly cautious when making decisions to avoid potential evidentiary dissonance (where a paradigm is supported by an apparent abundance of evidence, but with little basis in actual reported scientific findings and a 'too big to fail' complex; see discussion by Sheaves et al., 2020). This phenomenon has been reported in other fields of environmental management (Sheaves et al., 2020; Lemasson et al., 2023). Here, this might occur if we were to promote the paradigm that GGI is beneficial without establishing evidentiary scientific support first (Gauff et al., 2023).

GGI may still suffer from a lack of underpinning science that can be used to support evidence-based management. Firth et al. (2020) highlighted some of the limitations to the evidence base, including difficulty generalizing the findings (different metrics, timescales, environmental conditions, sampling protocols). Significant progress has been made in response to the call for action expressed in their 2020 paper (see Table 1 for summary; see also the work discussed in this paper). Importantly, many new schemes have emerged since 2020 that have been driven by the practitioner and not the scientific community (see CIRIA, in prep for comprehensive inventory of UK examples). Ecological responses from such practitioner-led projects will not be subjected to the same rigour and criticism as peer-reviewed scientific literature and may be difficult to find online. Crucial evidence may also be 'hidden' from the general scientific literature, their access restricted due to institutional or corporate confidentiality, or due to a lack of transparency or willingness to share from their authors (Sheaves et al. 2016). We must learn from other fields of environmental management where collaborative sharing across sectors is becoming more standard practice (such as in the field of oil and gas decommissioning; e.g., INSITE programme, Table 3).

The past few years have seen a clear effort to improve the GGI evidence base, but also to consolidate it, with evidence syntheses starting to emerge (Strain et al., 2018a; O'Shaughnessy et al., 2020a; Evans et al., 2021). These efforts are promising but become rapidly out of date to the rapidly evolving field and proliferation of the literature. Some fall somewhat short of the standards required of evidence syntheses due to not being based on robust, transparent, and repeatable methods. Others may not reflect accurately on the state-of-the-art, being selective of only a few specific GGI interventions, rather than encompassing all possible known GGI interventions, and only incorporate published literature, excluding the grey literature (thus likely biasing the results towards positive results and 'successes' rather than 'failures'). Future syntheses should thus aim to follow the gold standards (see Lemasson et al., 2021, 2022 for a systematic protocol and map of evidence available to inform oil and gas decommissioning), incorporate all possible GGI interventions, and include both published and grey literature where possible, to provide the best possible scientific support for management and decision making when it comes to applying GGI. If 'reefing' of oil and gas and offshore windfarms as a decommissioning option becomes more and more popular in the future (Schläppy et al., 2021), how these structures could be modified or treated prior to reefing to maximize benefits (such as biodiversity enhancement, or targeted species overexploited for instance) should be investigated (see Knights et al. 2023, 2024 for examples using expert scientific consensus). GGI could be applied to decommissioned offshore structures (either prospectively or retrospectively) to achieve greater ecological benefits. Whilst some studies do exist that touch on this topic for existing operational structures (Langhamer and Wilhelmsson, 2009; Langhamer, 2012; Bos et al., 2021; Roach et al., 2022), we highlight that this is a major knowledge gap that should be investigated for both existing and decommissioned structures in the future.

5. Concluding remarks

In this paper, we provide an update on state-of-the-art of GGI research since 2020 (summarised in Table 1). Researchers are designing better experiments by testing the independent and interactive effects of habitat area and spatial configuration, standardizing comparisons between units by using small sampling units or compartmentalizing data by microhabitats, considering greater numbers of materials, experimental units, surface orientations and context-specific refuge provision (see section 2.1). Researchers are also going beyond simple biodiversity metrics and capture other biological, environmental, and societal information (see section 2.2.). In many places, existing installations and experiments are being revisited to test responses over longer timescales and repeated and replicated under different environmental contexts (see section 2.3). Far greater attention is being given to reporting on unintended outcomes such as GGI performing as ecological traps, supporting invasive species and being inundated by sediments or indeed occasional happy accidents with unexpected benefits (see section 2.4). Whilst limited evidence is currently available on ‘real world’ scaled up installations (but see Living Seawalls, Box 4), some large scale projects are beginning to emerge (see section 2.5). Finally, appropriate policies and guidance are beginning to emerge (see section 2.6). Table 3 provides links to some of the major tools and resources that are available to practitioners and educators.

The global evidence base for GGI is rapidly expanding. We urge researchers and practitioners not to oversell their results and report all findings in a nuanced manner, reflective of the short-term duration and scale of interventions in the wider context of experimental damage caused by the construction, particularly if it is a new development. We recommend synthesizing this evidence base using systematic mapping approaches and learning from other fields of environmental management where collaborative sharing across sectors is becoming more standard practice. Importantly, some major knowledge gaps remain. Critically, properly scaled up examples of GGI are still lacking. Whilst the Living Seawall (Box 4) is a great example of how science is being put into practice at real world scales, the driver behind this example is still very much the research community and not the practitioner. Similarly, how such installations alter connectivity patterns across seascape scales remains unknown. Looking to the future, a major challenge will be to predict how GGI will interact with climate change for both NNS and native species that are on the move (Cannizzo et al., 2019, 2020). This is particularly true if future GGI efforts are applied to structures spanning natural barriers to dispersal and biogeographic provinces (e.g., Forbes’ Line in the UK and Ireland, Firth et al. 2021b; see also Lacroix and Pioch, 2011). Whilst great advances have been made through large-scale interdisciplinary projects, these are often spatially restricted to national or regional scales based on funding mechanisms. By addressing the above-mentioned challenges we will dramatically improve our ability to implement GGI in appropriate ways that can be classed as laurel wreaths over fig leaves or Trojan horses and achieve true win-wins for humanity and nature.

The 51 authors on this paper represent institutes in 14 countries spanning five continents. Here we argue the need for a paradigm shift of current funding strategies and research programmes that will encourage the development, implementation, and translation of GGI science at global scales. There is currently a lack of science and practice outside of academic sectors in the developed world, and insufficient global funding mechanisms that can support such collaborations. This rationale further evokes the need for equitable North-South partnerships in science informing GGI, that is well embedded in the UN Local2030 agenda, with a key focus on sharing of tools and demand-driven research and action. There is, thus, a collective need for schemes that encourage intersectoral and transsectoral research, knowledge exchange, and capacity building to optimize GGI in the pursuit of contributing to sustainable development.

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BOXES

Box 1. The Palm Effect

For millennia, people have been fascinated by the idea of artificial islands. In Neolithic Ireland and Scotland people lived on defensive wooden islands in lakes called crannógs (Wood-Martin, 1886). Construction of the ancient island site of Nan Madol, Micronesia began as early as the 8th century (Athens, 1980). Indeed, Thomas More's (1516) Utopia was an imagined artificial island in which everything was perfect; a concept that has transcended time with many recent artificial islands being designed and promoted as utopian paradises where the global elite can live, work and play in eco-friendly, green and smart cities (Grydehøj and Kelman, 2016). More recently, this has even gone as far as the concept of "seasteading"; with claims of how floating nations "will restore the environment, enrich the poor, cure the sick, and liberate humanity from politicians" (Quirk and Friedmann, 2017). Nowhere exemplifies this concept better than Dubai, with its myriad of artificial islands and where one can own their very own island in the shape of their country of choice.

Upwards of 479 artificial islands are known to exist (Bugnot et al., 2021), ranging in scope and design from simple rounded and fractal designs (Chee et al., 2017) to national symbols (e.g. The Pearl, Qatar), and even maps of the whole world (i.e., The World, Dubai). Incidentally, the construction of Dubai's The Universe was halted due to the 2008 global economic crash (El-Sheshtawy, 2010). Jackson and della Dora (2011) describe contemporary artificial islands as "metageographical terra-forms that normalize and naturalize people's expectations, knowledge and interactions with the world". Indeed, there is no doubt that the Palm Islands have inspired many of these designs through what we refer to as "The Palm Effect". The trend first caught on in the Arabian Gulf States but has since spread to other regions (Figure 5). For instance, the developers behind the plans for a megadevelopment on the Malaysian island of Langkawi were quoted to claim that the development "would be akin to that in Dubai", encouraging tourism and investment (The Star, 2018). Similarly, the developers behind Lagos' Eko Atlantic refer to it as "Africa's Dubai" (Eko Atlantic, 2020). Jackson and della Dora (2009) further point out that artificial islands are the 'new cultural icons', are a 'must' for aspiring 'global cities', or for countries at the fringes of capitalist economy striving to get international attention. All artificial islands will inevitably be low-lying and potentially vulnerable to future water intrusion by storms or sea level rise. In the face of pervasive global climate change, rising and stormier seas, and population increases, these hybrids of nature and technology are likely to become even more pervasive, perpetuating the Palm Effect.

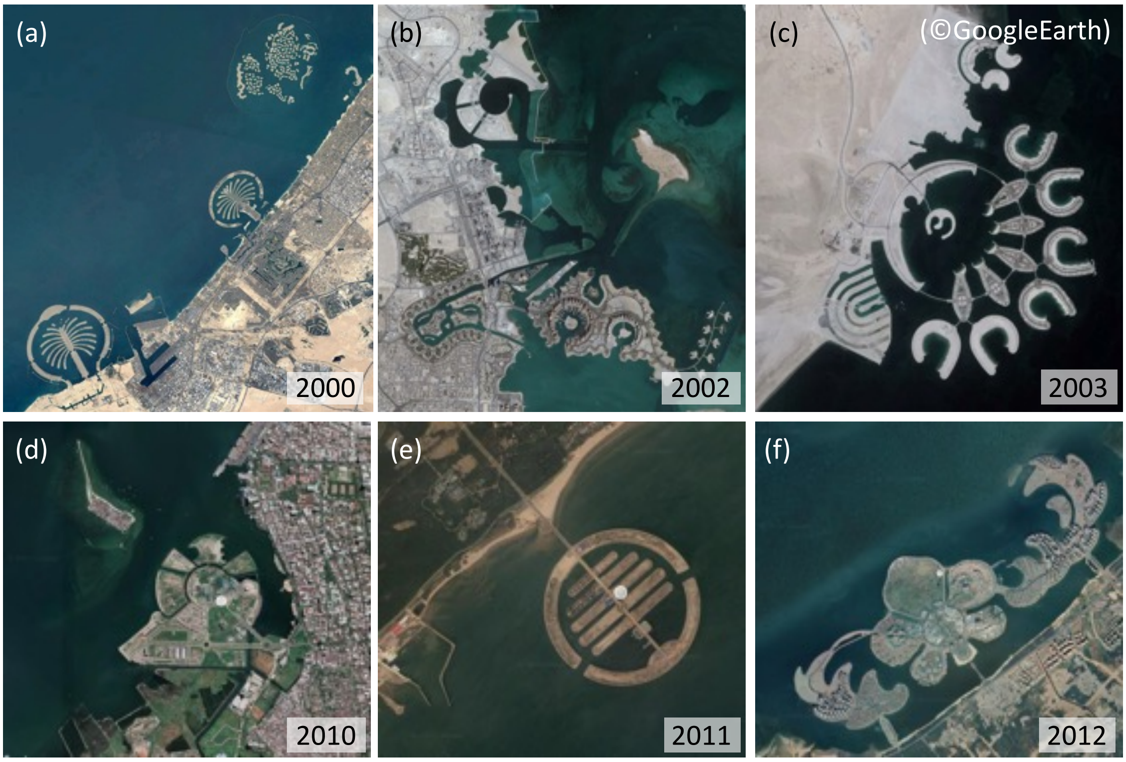

Figure 5. The Palm Effect – coastal artificialization on the Arabian Peninsula and Asia: (a) The Palms and The World, Dubai, United Arab Emirates; (b) The Pearl, Qatar; (c) Durrat al Bahrain, Bahrain; (d) Center Point of Indonesia (representing a Garuda (eagle), Sulawesi, Indonesia; (e) Qinhuangdao, Hebei, China; (f) Ocean Flower Island, Hainan, China. The numbers in the bottom right indicate the year that construction started (all images from Google Earth).

Box 2. Artificialization of the global coastline

Coastal development is the anthropogenic change in a seascape through the construction of artificial structures within sight of the coastline. Firth et al. (2020) stated that Asian and Middle Eastern countries have constructed some of the most ambitious and iconic land reclamation projects. Since then, several empirical large scale and even global studies have emerged that can attest to this. For instance, Bugnot et al. (2021) found that China was responsible for 40% of global construction in the marine environment, followed by South Korea (10% of global) and the Philippines (8% of global). Sengupta et al. (2020) found that since 1988, >700 km^2 of land reclamation has occurred in just eight Asian cities, with >35% of this happening in Shanghai alone. Similarly, Chee et al. (2023) found that since 1991, >82 km^2 has been or will be reclaimed in Malaysia. Penang Island, Malaysia is one of the most densely populated, and fastest growing urban settlements globally (Chee et al., 2017) (see Figure 1). Bounded by the Penang Strait to the east, and hills to the west, space for further development is severely limited. The solution to tackle this social problem has been to build upwards (i.e., construction of high-density, high-rise buildings) and outwards (i.e., land reclamation and the construction of five large artificial islands).

Firth et al. (2020) also suggested that coastal African countries that are experiencing rapid population growth are the most vulnerable to future habitat loss. Whilst Claassens et al. (2022) only found that 2.9% of the 3000-km South African coastline is armoured this number is likely to be much higher given the large number of informal settlements. Further armoring is likely in the future due to growing urban populations and additional pressure from Operation Phakisa to transform the South African coastal environment to unlock the 'blue economy' (Loureiro et al., 2022). With 10 million m2 of reclaimed land protected by an 8.5 km long seawall, Nigeria is currently undertaking one of the world's largest land reclamation and construction projects (Ajibade, 2017). Once complete, Eko Atlantic City will be the size of Manhattan (Eko Atlantic, 2022). Incidentally, whilst its name uses an old term for the Nigerian city of Lagos ('Eko') it may incite perceptions of eco-friendly development. There is potential for Eko Atlantic to act as a regional catalyst (see The Palm Effect, Box 1) for similar coastal megadevelopment in other rapidly growing African countries. Collectively, all this evidence is reinforcing the assertions by Firth et al. (2020) that continued artificialization of the coastline is inevitable, particularly in Africa and the Middle East.

In the first global estimation of the footprint of marine construction, Bugnot et al. (2021) reported that 32,000 km^2 of seafloor had been reclaimed. In their audit of artificial shoreline extent, Sempere-Valverde et al. (2023) estimated that ~16% of the global coastline was armoured; with most development in lagoons, estuaries and bays, and also in regions characterized by middle to high incomes. Many estimates of coastal armouring exist for developed nations (Table 2), but the figures are less robust for developing nations as many coastal settlements are 'informal' and coastal protection and land reclamation practices are haphazard, opportunistic, and unregulated (Palmer et al., 2010). Most of the predicted population growth across the globe by 2050 is expected to occur in low- and middle-income countries (Cohen, 2003; Nieves et al., 2017) and, by 2030, an estimated 38% of the global population will live in the near-coast zone (the area located <100 km from the coast and below 100 m elevation, Kummu et al., 2016). Whilst this zone only occupies 9% of global land area, it supports 62% of cities with over 5 million inhabitants (Firth et al., 2016a) and 42% of global Gross Domestic Product (Kummu et al., 2016). Many regions already claim that >50% of their coastlines are artificial (Table 2), and it seems inevitable that coastal armouring will continue apace.

Box 3. Examples of Trojan horses, Fig leaves and Laurel wreaths from artificial reefs, rigs-to-reefs and GGI of artificial structures.

Firth et al. (2020) highlighted a range of limitations and unknowns noting that GGI experiments could be assigned to one of three categories: Trojan horses, projects that cause environmental damage either through deliberate or misguided intent; Fig leaves, projects that merely cover up environmental damage caused by the development; and Laurel wreaths, win–win projects with measurable benefits for humans and nature. The authors suggested that much could be learnt from the artificial reef literature, a more established discipline subject to intense critical evaluation (e.g., Pioch et al., 2011; Castello y Tickell et al., 2019; Komyakova et al., 2021; Reis et al., 2021; Vivier et al., 2021b; Bartholomew et al., 2022; Salaün et al., 2022). Using artificial reefs, rigs-to-reefs (decommissioned oil and gas structures repurposed as artificial reefs), and GGI on coastal infrastructure, we provide illustrations of Trojan Horses, Fig Leaves and Laurel Wreaths.

ARTIFICIAL REEFS

LAUREL WREATH: Artificial reefs (ARs) can simultaneously benefit nature and humans. ARs can increase fish production, including commercially-important species, by providing refuge, breeding sites and increased feeding opportunities (Pondella et al., 2002; Cresson et al., 2014; Roa-Ureta et al., 2019; Folpp et al., 2020). ARs can provide additional hard-bottom habitat as attachment sites for commercially-important bivalves (Goeltz et al., 2020) and important habitat-forming species like macroalgae and corals (Campos et al., 2020, Higgins et al., 2022). ARs can be used to protect important habitats from trawling (Relini et al., 2007). ARs built for tourism and diving can also support diverse biota (Jackson et al., 2004; Smith et al., 2022) and relieve pressure on natural reefs (Belhassen et al., 2017; Firth et al., 2023).

FIG LEAF: An AR was created to compensate mudflat loss following creation of a dredged material disposal site. Burton et al. (2002) found that the annual secondary production/unit area from sessile invertebrates at the offset AR was 11–67 times higher than natural soft-sediment habitat, but total annual secondary production was 1.3–7.6 times lower. The authors concluded that the artificial reef improved secondary production, but not enough reef was created to fully offset the lost habitat. The Palm Jebel Ali was built on natural coral habitat, (Burt et al., 2008), and research found that coral diversity was lower and fish communities were different on breakwaters, although coral percent coverage was higher (Burt et al., 2009). ARs were created in Qatar as transplantation sites for coral that would be destroyed by a new oil pipeline, and initial coral survivorship results were promising (Deb et al., 2014). However, survivorship of transplanted coral is generally low, with only 20% of studies reporting survivorship over 90% (Boström-Einarsson et al., 2020).

TROJAN HORSE: Some ARs are simply ocean dumping (Chou, 1997), either inadvertently if the reef fails to support much life, or through the deliberate disposal of waste material at sea under the guise of 'reef creation' (e.g. The Osborne Reef, Florida, Figure 6). Scrap materials used as 'reefs' may have chemical pollutants associated with them (Dodrill et al., 2011; Gaylarde et al., 2021), and they may move during storms and impact natural habitats (Turpin and Bortone, 2002; Sherman and Spieler, 2006; Morley et al., 2008). ARs attract both fish and fishers, and can increase catch rates in the short term but contribute to regional overfishing in the long term (Watanuki and Gonzales, 2006; Simon et al., 2011). ARs can also facilitate the spread of invasive hard-bottom species (Adams et al., 2014; Airoldi et al., 2015).

RIGS-TO-REEFS

LAUREL WREATH: We do not think that there are examples of laurel wreaths when it comes to the decommissioning of oil and gas platforms as artificial reefs (so called "rigs-to-reefs"). The negative impacts of the placement of such large structures on the environment are too great for any compensatory efforts to be viewed as positive.

FIG LEAF: All retrospective rigs-to-reefs are fig leaves. It is undeniable that they can provide substantial habitat for a diversity of marine life (e.g., Gass and Roberts, 2006; Sammarco et al., 2012; Friedlander et al., 2014). However, this is often based on descriptive studies that have no formal comparison with natural habitats, and the perceived benefits are often weighted more towards humans than nature. For example, the developers save money from not having to remove the structure (Bressler and Bernstein, 2015), and fishing communities and marine recreationists may benefit from enhanced activities following decommissioning (e.g., Ajemian et al., 2015).

TROJAN HORSE: Rigs-to-reefs can become Trojan horses through a range of different pathways. For example: (1) a ”successfully” decommissioned rig that has been converted to a “reef” and continues to function as habitat and provide human recreational activities, can retrospectively cause further harm if it suffers a leak, leading to the release of oil into the environment; (2) assertions that offshore platforms support better fisheries than natural habitats (Claisse et al., 2014), may prospectively cause environmental damage through influencing governments to relax regulations or develop policies (e.g. the US National Fishing Enhancement Act (Public Law 98-623, Title II), BSSE, 2018) which may play a role in facilitating increased oil production and the proliferation of artificial structures on the seabed.

GREENING OF GREY INFRASTRUCTURE (GGI) ON MARINE ARTIFICIAL STRUCTURES

LAUREL WREATH: Disused docks that are redeveloped for commercial/residential purposes. As part of the process water quality needs to be improved. The encouragement of native mussels increased natural biofiltration which improved water quality (Figure 6c, Wilkinson et al., 1996) and led to colonisation of diverse assemblage of marine life (Allen et al., 1995; Hawkins et al., 1999; Firth et al., 2023b). Artificial water circulation further enhanced water movement and quality.

FIG LEAF: A boulder on a breakwater that is located in a sedimentary environment is replaced with a precast concrete habitat-enhancement unit (Firth et al., 2014). The unit encourages colonisation of greater diversity of reef taxa than the adjacent boulders. The enhancement of rocky-reef taxa is only compensation for the loss of sedimentary habitat and biodiversity in the footprint of the structure.

TROJAN HORSE: A shore-parallel breakwater is enhanced through the addition of novel habitats such as pits or rock pools (e.g. Evans et al., 2015). Hypothetically the habitats become dominated by invasive species that prevent establishment of native taxa. This could be exacerbated if the breakwater is outside of the known range for the invasive species, thus facilitating spread through the steppingstone concept (Mineur et al., 2012; Airoldi et al., 2015). This could equally be true if the eco-engineering interventions facilitated establishment and spread of parasites, pathogens, pests or other harmful organisms that have been demonstrated to be spread by artificial structures (e.g., Vila et al., 2001; Villareal et al., 2007; Lo et al., 2008; Ishii & Katsukoshi, 2010; Duarte et al., 2012). To date, no such reports have been made from eco-engineering trials, but we urge researchers to monitor eco-engineered habitats over longer timescales and report on any negative impacts. An even more serious example might be where a local government views a planning application for the construction of an artificial island more favourably based on the promise of the inclusion of eco-engineering. Even if the eco-engineering goals are perceived to be successful later down the line, it was partly responsible for the development getting approved which led to the destruction of natural coastal habitats.

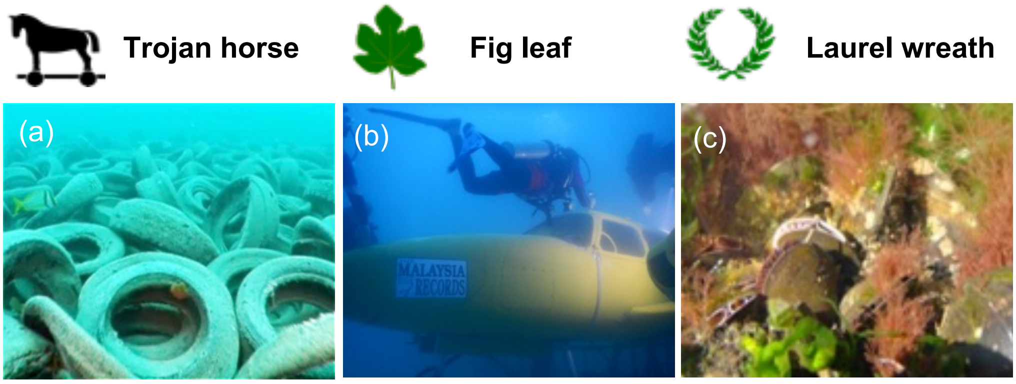

Figure 6. (a) Trojan horse: the Osborne reef constructed out of tyres, Florida, USA (Photo credit By Navy Combat Camera Dive Ex-East); (b) Fig leaf: the placement of disused plane in the sea as artificial reef, Malaysia (Photo credit Quek Yew Aun), (c) Laurel wreath: mussels supporting high biodiversity and filtering water on the walls of the Albert Dock, Liverpool, UK (Photo credit Louise Firth).

Box 4. Living Seawalls

Ecologists at the Sydney Institute of Marine Science have partnered with industrial designers from Reef Design Lab to develop Living Seawalls - a modular system by which critical habitats for marine life can be added to marine constructions at scale (Figure 7a). Using 3D printing technology, the pits, crevices and pools often present on natural shorelines are recreated on modular panels. The panels, which can be manufactured from upcycled materials, are then fitted to new or existing marine developments in customizable ‘Living Seawalls’ mosaics designed to last at least 20 years in locations of low wave exposure.

To date, ten different panel designs have been developed, each of which supports a distinct ecological community (Bishop et al., 2022). After only 2 years, Living Seawalls in Sydney Harbour have been colonized by over 100 species (see supplementary information in Bishop et al., 2022 for full details of the species found) and up to 35% more fish, seaweed and invertebrates than unmodified seawalls (Vozzo et al., 2021b).

Since the first Living Seawall was installed under the Sydney Harbour Bridge in late 2018, panels have been installed at 20 relatively sheltered sites in Australia and internationally in Plymouth, UK (Figure 7b), Gibraltar, Wales and Singapore, with installations also planned for Boston, USA. These include urban renewal projects spanning hundreds of meters and private or public water frontages of tens of meters or less. Uptake of Living Seawalls has been assisted by the team’s development of frameworks for ecologically enhancing marine infrastructure (Dafforn et al., 2015, 2016; Mayer-Pinto et al., 2017). These were used by the multinational company Lendlease in the Barangaroo urban renewal project and by the State Government of New South Wales in the Sydney Fish Markets redevelopment.

The project has also generated awareness of eco-friendly construction through public seminars and outreach events, stakeholder workshops, and social media. Living Seawalls has been featured in over 20 international print, television and radio pieces and displayed in seven leading design museums, globally. Living Seawalls was a finalist for the 2021 Earthshot Prize, the winner of the 2022 Banksia Foundation Biodiversity Award, and a Top Innovator in the World Economic Forum’s 2022 UpLink BiodiverCities challenge.

Figure 7. (a) The Sydney Living Seawall. Image on right is showing close up of the water retaining panel (Photo credits Living Seawalls.com.au). (b) Green algae dominating early successional stages of the Living Seawalls in Plymouth only three months after installation in August 2023. Image on the right shows the first limpet to colonise the panels (November 2023). It is expected that once limpets arrive, they will graze down the algae making space for other species to colonise (Photo credits Aeden Cooper).

Tables

Table 1. Summary of criticisms and suggested actions made by Firth et al. (2020) coupled with references to recent advances and progress on these criticisms.

Limitations/unknowns	Suggested actions	Key papers and recent advances/progress
Experiments typically consider limited numbers of treatments/material types, ignore differences in surface area and are small scale (i.e. few meters) and short duration (e.g. less than 12 months).	Researchers should design better experiments (see section 2.1).	Testing the independent and interactive effects of habitat area and spatial configuration (Loke et al., 2019a,b). -Standardizing comparisons between units by using small sampling units or compartmentalizing data by microhabitats (Strain et al., 2021; Bishop et al., 2022). -Consideration of greater number of materials (Becker et al., 2020; Hsiung et al., 2020; Schaefer et al., 2020, 2023; Vivier et al., 2021a; Natanzi et al., 2021; Dauvin et al., 2022; Dodds et al., 2022; Hartanto et al., 2022; Lapinski et al., 2022; Summers et al., 2022). -Consideration of larger number of experimental units (Bishop et al., 2022). -Consideration for context-specific refuge provision (Strain et al., 2021; Aguilera et al., 2023b). -Consideration of variability with orientation differences (Grasselli and Airoldi, 2021).
Limited metrics measured.	Go beyond simple biodiversity metrics and capture other biological, environmental, and societal information (see section 2.2).	-Consideration of more biodiversity metrics (Levrel et al., 2012; Bas et al., 2016; O'Shaughnessy et al., 2023). -Consideration of how GGI can moderate stressors (Bradford et al., 2020; O'Sullivan et al., 2020; Rouse et al., 2020; Sedano et al., 2020a; Strain et al., 2020; Pioch and Souche, 2021; Vozzo et al., 2021a; Dodds, 2022). -Consideration of ecosystem functioning metrics such as biofiltration (Malerba et al., 2019; Riascos et al., 2020; Rouse et al., 2020; Vozzo et al., 2021a; Bulleri et al., 2022; Raoux et al., 2022; Mayer-Pinto et al., 2023). -Considering socio-economic benefits (Morris et al., 2016; Evans et al., 2017; Kienker et al., 2018; Evans et al., 2019; Morris et al., 2019; Strain et al., 2019; Sawyer et al., 2020; Fairchild et al., 2022; Accola et al., 2022a,b; Salaün et al., 2022; Aguilera et al., 2023b). -Consideration of other metrics such as trophic structure (Espinosa et al., 2020; Sedano et al., 2020a,b; Raoux et al., 2022), sedimentation (Bone et al., 2022b; Martinez et al., 2022), transport of wrack (Strain et al., 2018b; Critchley et al., 2021), litter accumulation (Aguilera et al., 2016, 2018, 2023a), use by terrestrial predators or pest species (Aguilera et al., 2023a), early life stage usage (Mafanya, 2020; Reddy, 2022).

Uncertainty on how GGI will perform under different environmental scenarios.	Repeat, revisit, replicate experiments (see section 2.3).	-Replication across environmental contexts (Hsiung et al., 2020; Strain et al., 2021; Chee et al., 2021; Clifton et al., 2022; O'Shaughnessy et al., 2021) and changing environmental conditions (Sun et al., 2022; Waltham et al., in review). -Consideration of correlates that may interact with GGI interventions (Strain et al., 2018a; Clifton et al., 2022; Strain et al., 2021; O'Shaughnessy et al., 2021; Bishop et al., 2022; Clifton et al., 2022). -Increasing duration of experiments (Martins et al., 2016; Bender et al., 2020, Chee et al., 2020, Bishop et al., 2022).
Unintended outcomes such as whether GGI facilitates spread of invasive species or acts as an ecological trap or environmental/biological filter.	Report on unintended outcomes (both positive and negative) (see section 2.4).	-Consideration of ecological traps (Swearer et al., 2021; Komyakova et al., 2021, 2022). -Consideration of happy accidents in relation to endangered species (Claassens, 2016; Claassens and Hodgson, 2018; Claassens et al., 2018; Ostalé-Valriberas et al., 2022) -Invasive species (Wells and Bieler, 2020; O'Shaughnessy et al., 2021; Adams et al., 2021; Vozzo et al., 2021a; Gauff et al., 2023). -Sedimentation (Bone et al., 2022b).
Dearth of large demonstration tests that show how interventions will perform when scaled-up operationally in 'real' developments.	Research and practice should move together in tandem (see section 2.5).	Limited progress in this area, but see Box 4 on Living Seawalls. -See recent papers on Seattle Seawall Project (Sawyer et al., 2020; Accola et al., 2022a,b). -See CIRIA, in prep for inventory of some large-scale examples of scaled-up GGI installations in the UK.
Limited guidance for practitioners	Develop appropriate policies and guidance (see section 2.6).	-Co-creation of guidance, case studies and policy instrument with practitioners (Naylor et al., 2012, 2017; Chee et al., 2021b; Lai et al., 2022; CIRIA, 2024). -Document levers that could support GGI (Naylor et al., 2012, Pioch et al., 2018; Evans et al., 2019; Mayer-Pinto et al., 2017; Claassens et al., 2022). -Identify policy gaps (RSA 2014; GIBOP, 2019).

Table 2. Summary of studies charting the extent of armoured coastline on national and regional scales.

(a) International/national			
Region	**Location**	**% Artificial**	**Source**
Global	Total	16	Sempere-Valverde et al., 2023
Europe	Total	6.4	Eurosion, 2004
	Belgium	34	Eurosion, 2004
	Belgium	85	Gregory, 2010
	Bulgaria	10	Stancheva et al., 2011
	Cyprus	20	Eurosion, 2004
	Denmark	12	Eurosion, 2004
	England	38	Gregory, 2010
	England	46	Masselink and Russell, 2013
	England	50	Turner et al., 2007
	Estonia	1	Eurosion, 2004
	Finland	1	Eurosion, 2004
	France	15	Eurosion, 2004
	Germany	18	Eurosion, 2004
	Greece	4	Eurosion, 2004
	Ireland	3	Eurosion, 2004
	Ireland	8	Masselink and Russell, 2013
	Italy	8	Eurosion, 2004
	Latvia	3	Eurosion, 2004
	Lithuania	12	Eurosion, 2004
	Malta	7	Eurosion, 2004
	Monaco	89	www.medam.org
	Northern Ireland	20	Masselink and Russell, 2013
	Northern Ireland	15	Gibson 2011
	Northern Ireland	32	Cooper and Jackson, 2018
	Poland	3	Eurosion, 2004
	Portugal	5	Eurosion, 2004
	Scotland	7	Masselink and Russell, 2013
	Slovenia	18	Eurosion, 2004
	Spain	10	Eurosion, 2004
	Sweden	1	Eurosion, 2004
	The Netherlands	60	Eurosion, 2004

	United Kingdom (total)	5	Eurosion, 2004
	United Kingdom (total)	18	Masselink and Russell, 2013
	Wales	28	Masselink and Russell, 2013
Africa	South Africa	2.9	Claassens et al., 2022
Asia	Singapore	63	Lai et al., 2015; Loke et al., 2016
	Malaysia	1	Sa and Boon, 2010
	Japan	27	Koike, 1996
	Japan	34	Koike, 2003
	China	60	Guan, 2013
Americas	USA (total)	9	NOAA, 2012
	USA (total)	14	Gittman et al., 2015
	Chile	6.6	Aguilera, 2018
(b) Regional and Local			
Region	**Location**	**% Artificial**	**Source**
Middle	Dubai, United Arab Emirates	130	Burt et al., 2013
Australasia	Great Barrier Reef, Australia	9.4	Waltham and Sheaves, 2015
	Adelaide, Australia	53	Floerl et al., 2021
	Gladstone, Australia	23	Floerl et al., 2021
	Darwin, Australia	10	Floerl et al., 2021
	Hobart, Australia	22	Floerl et al., 2021
	Geelong, Australia	56	Floerl et al., 2021
	Sydney Harbour, Australia	50	Chapman, 2003
	Sydney Harbour, Australia	85	Strain et al., 2019
	Port Phillip Bay, Melbourne, Australia	25	Strain et al., 2019
	Various estuaries in New South Wales, Australia	3.8-46	Creese et al., 2009
	Derwent Estuary, Hobart, Tasmania, Australia	19.2	Strain et al., 2019
	Waitemata Harbour, Auckland, New Zealand	20.6	Strain et al., 2019
	Auckland, New Zealand	47	Floerl et al., 2021
	Dunedin, New Zealand	77	Floerl et al., 2021
	Tauranga, New Zealand	20	Floerl et al., 2021
	Wellington, New Zealand	77	Floerl et al., 2021
	Whangarei, New Zealand	25	Floerl et al., 2021
	Nelson, New Zealand	50	Floerl et al., 2021
	Bluff, New Zealand	21	Floerl et al., 2021
	Lyttelton, New Zealand	25	Floerl et al., 2021

	Picton, New Zealand	71	Floerl et al., 2021
	Napier, New Zealand	93	Floerl et al., 2021
	Westport, New Zealand	51	Floerl et al., 2021
	New Plymouth, New Zealand	65	Floerl et al., 2021
	Opua, New Zealand	5	Floerl et al., 2021
	Greymouth, New Zealand	65	Floerl et al., 2021
Europe	Emilia Romagna, Italy	60	Bacchiocchi and Airoldi, 2003
	Belfast, UK	88	Floerl et al., 2021
	Ravenna Port, Italy	70	Strain et al., 2019
	Varna Bay, Bulgaria	10	Stancheva et al., 2011
	Plymouth Sound, UK	33	Knights et al., 2016
	French Mediterranean	11	www.medam.org
	Weser Estuary, Germany	60	Wetzel et al., 2014
	Santander Harbour, Spain	42	Strain et al., 2019
	Galway Bay, Ireland	7	Firth et al., 2016b
	Galway City	45	Firth et al., 2016b
	Cardiff, UK	87	Floerl et al., 2021
	Edinburgh, UK	89	Floerl et al., 2021
Africa	eThekwini, South Africa	11	Corbella and Stretch, 2012
Asia	Zhoushan Islands, China	65	Zhang et al., 2014
	Xiamen Harbour, China	90	Strain et al., 2019
	Victoria Harbour, Hong Kong	95	Lam et al., 2009
	Penang Island (east coast), Malaysia	88	Chee et al., 2017
	Balneário Camboriú, Brazil	36	Piatto and Polette, 2012
	Okinawa Island, Japan	63	Masucci and Reimer, 2019
	Keelung Harbour, Taiwan	80	Strain et al., 2019
Americas	Bermuda	15	Meyer et al., 2015
	Arraial do Cabo Harbour, Brazil	5	Strain et al., 2019
	Cartagena, Columbia	12	Stancheva et al., 2011
	Guam	15	NOAA, 2012
	American Samoa	7	NOAA, 2012
	Alabama, USA	11	NOAA, 2012
	Alaska, USA	2	NOAA, 2012
	California, USA	12	Griggs, 1998
	California, USA	14	NOAA, 2012

	County comparisons, California, USA	0.5-61	Hanak and Moreno, 2012
	Californian cities of Long Beach, Seal Beach,	>70	Dugan et al., 2011
	Connecticut, USA	18	NOAA, 2012
	New Haven, Connecticut, USA	74	Floerl et al., 2021
	Delaware, USA	12	NOAA, 2012
	Florida, USA	21	Florida DEP, 1990
	Florida, USA	20	NOAA, 2012
	Georgia, USA	1	NOAA, 2012
	Hawaii, USA	11	NOAA, 2012
	Oahu, Hawaii, USA	26	Campbell et al., 1988
	Louisiana, USA	3	NOAA, 2012
	Maryland, USA	15	NOAA, 2012
	Baltimore, Maryland, USA	71	Floerl et al., 2021
	Massachusetts, USA	11	NOAA, 2012
	Boston Harbour, Massachussetts,	58	Strain et al., 2019
	Mississippi, USA	12	NOAA, 2012
	New Hampshire, USA	4	NOAA, 2012
	New Jersey, USA	17	NOAA, 2012
	New Jersey, USA	17	Lathrop and Love, 2007
	New York, USA	24	NOAA, 2012
	North Carolina, USA	8	NOAA, 2012
	Oregon, USA	6	Dugan et al., 2011
	Oregon, USA	4	NOAA, 2012
	Portland, Oregon, USA	77	Floerl et al., 2021
	Pennsylvania, USA	36	NOAA, 2012
	Puerto Rico, USA	10	NOAA, 2012
	Rhode Island, USA	14	NOAA, 2012
	Providence, Rhode Island, USA	62	Floerl et al., 2021
	South Carolina, USA	1	NOAA, 2012
	Texas, USA	15	NOAA, 2012
	Virginia, USA	8	NOAA, 2012
	Norfolk, Virginia, USA	56	Floerl et al., 2021
	Virgin Islands, USA	8	NOAA, 2012
	Washington, USA	6	NOAA, 2012
	Duwamish Estuary, Washington, USA	66	Morley et al., 2012

	Puget sound, Washington, USA	30	Dugan et al., 2011
	Vancouver, Canada	75	Floerl et al., 2021
	Nanaimo, Canada	34	Floerl et al., 2021
	Halifax, Nova Scotia, Canada	59	Floerl et al., 2021

Table 3. Summary of major resources (reviews/reports, outreach and education and other outputs) for practitioners.

Description	**Source**
MAJOR REVIEWS/REPORTS	
Evans et al. (2021). Conservation Evidence Synopsis on biodiversity enhancement on marine artificial structures.	https://www.conservationevidence.com/data/index?synopsis_id%5B%5D=44
Naylor et al. (2017). A Framework for Integrated Green Grey Infrastructure (IGGI) (2017)	https://eprints.gla.ac.uk/150672/
ICE Manual of Blue-Green Infrastructure (2023). An essential reference for practicing engineers, masterplanners, landscape architects, water design specialists, ecologists, development companies, and local planning authorities	www.icebookshop.com
Pioch et Souche (2021). Eco-design of Marine Infrastructures: Towards Ecologically-informed Coastal and Ocean Development	https://www.wiley.com/en-us/Eco+design+of+Marine+Infrastructures:+Towards+Ecologically+informed+Coastal+and+Ocean+Development-p-9781119865957
Pioch et al. (2018). A literature review about coastal infrastructures eco-design (GGI), at micro- to macro-biological scales, factors involved in concrete-biota interactions and civil engineering	https://www.sciencedirect.com/science/article/abs/pii/S0925857418301976

recommendations	
OTHER SOURCES FOR PRACTITIONERS	
Institute of Civil Engineers 'Tech Talk' on GGI	https://vimeo.com/717607188
Oxford Bibliography feature on Ocean Sprawl	https://www.oxfordbibliographies.com/display/document/obo-9780199830060/obo-9780199830060-0237.xml
Book on Environmental Engineering, John Wiley & Sons ed.	https://www.wiley.com/en-ag/Eco+design+of+Marine+Infrastructures:+Towards+Ecologically+informed+Coastal+and+Ocean+Development-p-9781119865957
International Coral Reef Initiative (ICRI) on GGI (2017)	https://icriforum.org/documents/eco-designed-mooring-for-coral-reef-restoration/
Lai et al. (2022). A Guide to Implementing Coastal Nature Based Solutions for Singapore.	https://www.clc.gov.sg/docs/default-source/books/nbs-guide.pdf
SOURCES FOR EDUCATION & OUTREACH	
Biological Sciences Review on GGI (2023)	https://www.hoddereducation.co.uk/subjects/science/products/16-18/biological-sciences-review
Science Journal for Kids article on GGI (2016)	https://sciencejournalforkids.org/wp-content/uploads/2019/08/rockpools_article.pdf
The Conversation article on hybrid GGI (2021)	https://theconversation.com/a-20-foot-sea-wall-wont-save-miami-how-living-structures-can-help-protect-the-coast-and-keep-the-paradise-vibe-165076
The Conversation article on GGI (2018)	https://theconversation.com/future-ocean-cities-need-green-engineering-above-and-below-the-waterline-93843
Hakai Magazine article on GGI (2019)	https://hakaimagazine.com/features/fish-below-your-feet-and-other-solutions-for-a-living-harbor/