Enhancing the Biodiversity of Marine Artificial Structures

Global evidence for the effects of interventions

Ally J. Evans, Pippa J. Moore, Louise B. Firth, Rebecca K. Smith & William J. Sutherland

CONSERVATION EVIDENCE SERIES SYNOPSES

Enhancing the Biodiversity of Marine Artificial Structures

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Conservation Evidence Series Synopses

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Digital material and resources associated with this synopsis are available at

https://www.conservationevidence.com/

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Advisory Board

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1. About this book

1.1. The Conservation Evidence project

The Conservation Evidence project is constituted of four main parts:

1. The **synopses** of the evidence captured for the conservation of particular species groups or habitats, such as this synopsis. Synopses bring together the evidence for each possible intervention that was identified. They are freely available online and, in some cases, available to purchase in printed book form.

2. An ever-expanding **database of summaries** of previously published scientific papers, reports, reviews or systematic reviews that document the effects of interventions. This resource comprises over 7,800 pieces of evidence, all available in a searchable database on the website www.conservationevidence.com.

3. ***What Works in Conservation,*** which is an assessment of the effectiveness of interventions by expert panels, based on the collated evidence for each intervention for each species group or habitat covered by the synopses. This is available as part of the searchable database and is published as an updated book edition each year (https://www.conservationevidence.com/content/page/79).

4. An online **open access journal, *Conservation Evidence*** publishes new pieces of research on the effects of conservation management interventions. All the papers published are written by, or in conjunction with, those who carried out the conservation work and include some monitoring of its effects (https://conservationevidencejournal.com/).

You can learn more about the Conservation Evidence project and the methods behind it in Sutherland *et al.* (2019).

1.2. *The purpose of Conservation Evidence synopses*

Conservation Evidence synopses **do**	Conservation Evidence synopses **do not**
• Bring together scientific evidence captured by the Conservation Evidence project (over 7,800 studies so far) on the effects of interventions to conserve and restore biodiversity	• Include evidence on the basic ecology of species or habitats, or threats to them
• List all realistic interventions for the species group or habitat in question, regardless of how much evidence for their effects is available	• Make any attempt to weight or prioritize interventions according to their importance or the size of their effects
• Describe each piece of evidence, including methods, as clearly as possible, allowing readers to assess the quality of evidence	• Weight or numerically evaluate the evidence according to its quality
• Work in partnership with conservation practitioners, policy makers, and scientists to develop the list of interventions and ensure we have covered the most important literature	• Provide recommendations for conservation problems, but instead provide scientific information to help with decision-making

1.3. *Who is this synopsis for?*

If you are reading this, we hope you are someone who has to or wants to make decisions about how best to support, manage, and conserve the marine environment and its biodiversity. Specifically, someone who is taking action to enhance the biodiversity of marine artificial structures. You might be a marine advisor or consultant in the public or private sector, a landowner, developer or engineer, a marine conservationist, a campaigner, a policy maker, a researcher, or a concerned citizen. This synopsis summarizes scientific evidence relevant to your conservation objectives and the actions you could take to achieve them.

We do not aim to make your decisions for you, but to support your decision-making by telling you what evidence there is (or isn't) about the effects that your or others' planned actions could have on biodiversity. Here, by 'evidence', we mean any scientific studies found during our literature searches (see below section 1.6) that quantitatively reported the effects of conservation actions (interventions).

When decisions have to be made with particularly important or irreversible consequences, we recommend carrying out a systematic review, as the latter is likely to be more comprehensive than the summary of evidence presented here. Guidance on how to carry out systematic reviews can be found from the Centre for Evidence-Based Conservation at Bangor University (www.cebc.bangor.ac.uk).

1.4. Background

The marine environment is highly biodiverse, and this biodiversity provides essential goods and services to humans and enhances human well-being (Gamfeldt *et al*. 2015). Marine biodiversity is, however, facing multiple threats from human activities and human-induced climate change (Lotze *et al*. 2018). There is therefore an increasing need for evidence-based management and conservation of the marine environment and of all organisms that live in it.

Ocean sprawl is one major threat to marine biodiversity worldwide (Bugnot *et al.* 2021). As coastlines become increasingly developed and marine industries spread into offshore waters, artificial structures replace natural intertidal and subtidal marine habitats, with knock-on effects to the environment at local and regional scales. Impacts on the receiving environment include destruction of natural habitats within their physical footprints, altered hydrodynamic processes, and provision of new hard substrate for marine life to colonize, allowing species to spread into new areas, including non-native invasive species (Bishop *et al.* 2017; Heery *et al.* 2017; Mineur *et al.* 2012). Policy makers and environmental managers need to assess the likely impacts of new development proposals and to recommend and implement measures that avoid, minimize, restore/rehabilitate and/or compensate for them. 'Ecological engineering' (eco-engineering) or 'integrated greening of grey infrastructure' (IGGI; see Naylor *et al.* 2017) approaches have emerged to demonstrate how marine artificial structures can be enhanced to provide biodiversity benefits whilst simultaneously serving their primary engineering function. These habitat enhancement approaches are rooted in decades of observation and experimentation on rocky shores and reefs (reviewed in Hawkins *et al.* 2020). Artificial structures were used as simplified model systems for testing hypotheses about the relationships between habitats, ecological processes and biodiversity (e.g. Hawkins & Hartnoll 1982; Jones & Kain 1967; McGuinness & Underwood 1986). Observations and experimental outcomes prompted the design and testing of conservation interventions, and applied work in this field began in the early 2000s with the Europe-wide projects, Delos ("Environmental Design of Low Crested Coastal Defence Structures", EVK3-CT-2000-00041) and Theseus ("Innovative technologies for safer European coasts in a changing climate", FP7.2009-1, Contract 244104), leading to early guidance handbooks (Burcharth *et al.* 2007; Naylor *et al.* 2011). The concept of eco-engineering/IGGI is now becoming popular in some parts of the world and such approaches may contribute towards sustainable development ambitions (Evans *et al.* 2019). It is crucial, however, to ensure full transparency regarding the benefits that can (and cannot) be achieved, and to acknowledge that there is likely to be a net environmental impact of introducing artificial structures to the marine environment, regardless of actions taken to enhance their biodiversity (Firth *et al.* 2020).

When planning or reviewing proposals that involve placing new artificial structures in the marine environment, or maintaining, modifying or decommissioning existing ones, practitioners should scrutinize the available scientific evidence base for actions that can be taken to deliver biodiversity benefits and their likely effects. Although eco-engineering and IGGI in the marine environment is relatively new in practice, the evidence base has grown rapidly in recent years (Strain *et al.* 2018). Reviewing the evidence is a time-consuming and costly exercise, and inefficient when done repeatedly by many individuals on a case-by-case basis. To improve the accessibility of evidence, and the efficiency and consistency of evidence-based decision-making, this synopsis summarizes the available global scientific evidence of the effectiveness of conservation interventions for enhancing the biodiversity of marine artificial structures. The methods used to create it are outlined below.

1.5. Scope of this synopsis

1.5.1 Review subject

This synopsis covers published evidence for the effects of global conservation interventions aimed at enhancing the biodiversity of marine artificial structures. It includes both intertidal and subtidal structures built or placed along coastlines (including in estuaries) and offshore, on the seabed and in the water column. A list of terms used to describe structures is included in a Glossary (Appendix 1). It does not include evidence from the substantial literature on artificial reefs, reef restoration, laboratory-based studies, studies on biofouling reduction, comparisons of different designs commonly used in artificial structures where there is no obvious choice for conservation reasons (e.g. concrete vs steel structures), or pure ecological investigations. Instead, it focuses on tests of *in situ* conservation actions to enhance the biodiversity of structures that are engineered to fulfil a primary function other than providing artificial habitats. Nevertheless, this wider literature may contain useful transferable evidence and is highlighted, where applicable, in the background sections at the beginning of each intervention chapter. Studies that report the effects of natural reef restoration and creating artificial reefs for conservation are or will be covered in alternative synopses (e.g. see the Subtidal Benthic Invertebrate Conservation and the Marine Fish Conservation synopses online). Some interventions tested out-of-context (i.e. not strictly *in situ*), such as tests of environmentally-sensitive materials deployed for convenience on natural rocky reefs, have been included where the authors explicitly draw the link to the subject of this synopsis. However, such evidence should be considered with caution since the contexts in which artificial structures are built can have overarching effects on their biodiversity (Becker *et al.* 2021; Connell 2000; Ferrario *et al.* 2016; Perrett *et al.* 2006).

Enhancing biodiversity *per se* may not always be the goal of eco-engineering/IGGI practice. Objectives may instead be to support certain ecosystem functions and services (e.g. attracting filter-feeders to improve water quality), to promote commercial or subsistence fisheries, or to exclude non-native or nuisance species. These objectives are all, however, underpinned by certain species (or groups of species) that make up the biodiversity colonising structures. Effects of interventions on marine macroalgae, microalgae, invertebrates and fishes on and around structures are reported. However, effects on the wider receiving environment are not, since these are inherently difficult to measure and not often reported.

Decisions regarding soft or hybrid (i.e. a mixture of soft and hard) coastal management options as an alternative to building hard artificial structures are outside the scope of this synopsis (e.g. see Bridges *et al.* 2021). It is widely-accepted that soft and hybrid approaches are more sustainable, cost-effective and ecologically sound options for flood and coastal erosion risk management (Temmerman *et al.* 2013). Nevertheless, where hard artificial structures are considered appropriate and necessary, opportunities can be taken to enhance their biodiversity through the eco-engineering/IGGI actions considered here. We do not provide any information on the costs of interventions or their effects on the engineering function of structures, since these topics are currently outside the scope of Conservation Evidence. Some indicative costs of similar eco-engineering/IGGI approaches are presented in Naylor *et al.* (2017) and O'Shaughnessy *et al.* (2020), including the likely economies of scale when interventions are scaled up from research or pilot projects to commercial practice (Naylor *et al.* 2017). Naylor *et al.* (2017) assessed the likely effects of interventions on the engineering integrity and function of structures and concluded that there was no evidence that any of the interventions considered compromised structures. In fact, there is some evidence to suggest that marine organisms can provide a protective layer over engineering materials, creating a more stable microclimate at the surface, which can reduce weathering processes in intertidal environments (Coombes *et al.* 2013; 2017). Furthermore, flume experiments have shown that adding rough elements to plain seawalls to mimic bolt-on eco-engineering designs can reduce wave overtopping, thereby improving their engineering function (Salauddin *et al.* 2021). This is not considered further in this synopsis, but we do report the degree of failure of interventions when reported in the original studies (e.g. *x* of *y* created habitats fell off structures or were buried by sediments). Finally, this synopsis does not take into account the relative carbon (or wider environmental) footprint of implementing the various interventions described, which may not be trivial, particularly when concrete is used to manufacture intervention designs. Life cycle assessments should be undertaken to understand the likely net environmental impact of interventions under different scenarios (Heery *et al.* 2020).

1.5.2 Advisory Board

An advisory board made up of international academics and practitioners with expertise in marine management, conservation, landscape design, marine engineering, biogeomorphology and/or eco-engineering of marine artificial structures was formed. These experts provided input into the evidence synthesis at one or both of two key stages: a) developing a comprehensive list of conservation interventions for review and b) reviewing the draft evidence synthesis. The advisory board is listed above and online (www.conservationevidence.com/content/page/119).

1.5.3 Creating the list of interventions

At the start of the project, a comprehensive list of interventions was developed by scanning the literature and in partnership with the advisory board. A recent meta-analysis of eco-engineering interventions (Strain *et al.* 2018) provided a valuable starting point for describing and defining interventions. The list was also checked by Conservation Evidence to ensure that it followed the standard Conservation Evidence structure (described below). The aim was to include all actions that have been carried out or advised to enhance the biodiversity of marine artificial structures within the scope outlined above, whether evidence for the effectiveness of an action is available or not. During the synthesis process, further interventions were discovered and integrated into the synopsis structure, while those that fell outside the scope were removed. The final list of interventions and their definitions are listed in Appendix 2.

Unlike other Conservation Evidence synopses, we have not organized the list of interventions into categories based on the International Union for the Conservation of Nature (IUCN) classifications of threats (https://www.iucnredlist.org/resources/threat-classification-scheme) and convservation actions (https://www.iucnredlist.org/resources/conservation-actions-classification-scheme). Marine artificial structures are built for a variety of reasons, falling under different threat categories. Conservation interventions to enhance their biodiversity would fall under the ‘Land/water management’ or the ‘Species management’ action categories.

In total, we found 43 conservation interventions that could be carried out to enhance the biodiversity of marine artificial structures. The list of interventions was organized into those that can be applied to intertidal artificial structures (22 interventions; Chapter 2) and those that can be applied to subtidal ones (21 interventions; Chapter 3). We found evidence for the effects of 33 of these interventions. The evidence was reported as 176 summaries (118 for intertidal structures; 58 for subtidal structures) from 86 relevant publications found during our searches (see Methods below).

1.6. Methods

1.6.1 Literature searches

Because of the unique nature of this synopsis topic, we did not use the standard Conservation Evidence search methodology of subject-wide evidence synthesis (Sutherland *et al.* 2019). Instead, we used the following four methods to search for studies to include in this synopsis:

(1) We used keywords (Table 1) to search the Web of Knowledge (Science) database (www.webofknowledge.com). We selected keywords to describe: (i) our preliminary intervention list; (ii) marine artificial structures to which they can be applied; and (iii) the environmental contexts in which they can be applied. We did not restrict the timespan or publication indexes searched. Search results were obtained up to and including 4 August 2021.

(2) We searched two recent eco-engineering reviews by Strain *et al.* (2018) and O'Shaughnessy *et al.* (2020), and extracted all reviewed references within.

(3) We searched the online database www.conservationevidence.com for relevant publications that have already been summarized by the Conservation Evidence project.

(4) We asked the advisory panel to suggest any important studies that we might not capture using the methods above at the beginning of the process, then asked them if we had missed any again at the end.

Table 1. Keywords used to search for literature in Web of Knowledge. We used three "Topic" fields ("Topic" searches include titles, abstracts and keywords) joined with the "AND" operator (i.e. Topic 1 AND Topic 2 AND Topic 3). Search results, therefore, contained at least one of the terms included in each of the columns (i)–(iii). The * symbol represents any number of letters not specified. The " " symbols restrict the search to the exact phrases enclosed.

Keyword search strings		
(i) Topic 1 - interventions	**(ii) Topic 2 - structures**	**(iii) Topic 3 - context**
bioblock OR "bio* concrete*" OR "bio* enhanc*" OR canopies OR canopy OR cavities OR cavity OR coir OR complex OR crevice OR cut OR "eco* concrete*" OR "eco* engineer*" OR "eco* enhanc*" OR elevation OR epoxy OR fissure OR flowerpot OR glue OR "green concrete*" OR groove OR "habitat enhanc*" OR habitat-forming OR hole OR invasive OR IGGI OR INNS OR "greening grey" OR "greening the grey" OR ledge OR "marine concrete*" OR microhabitat OR mimic OR non-native OR "nuisance species" OR pit OR plant OR pool OR protrusion OR raise OR relief OR ribbon OR ridge OR "rock pool*" OR rockpool OR rope OR roughness OR seed OR "settlement plate*" OR slope OR "soft structure*" OR swimthrough OR "swim through*" OR swim-through OR texture OR "tidal pool*" OR "tide pool*" OR tile OR topography OR tower OR transplant	"armoured shore*" OR "artificial habitat*" OR "artificial infrastructure*" OR "artificial reef*" OR "artificial shore*" OR "artificial structure*" OR "manmade structure*" OR "man-made structure*" OR "ocean sprawl"	Coastal OR estuarine OR intertidal OR harbour OR marine OR offshore OR subtidal

Evidence from all around the world was included. However, only English language publications were included. Where a systematic review was found for an intervention, if the intervention had a small body of literature (<20 papers), all publications including the systematic review were summarized individually. If the intervention had a large body of literature (≥20 papers), then only the systematic review was summarized as were any publications published since the review or not included within it. Where a non-systematic review (or editorial, synthesis, preface, introduction, etc.) was found for an intervention, all relevant publications referenced within it were included, but the review itself was not summarized. However, if the review also provided new/collective data, then the review itself was also summarized (indicating which other summarized publications it included). Relevant publications cited in other publications summarized for the synopsis were not included (due to time restrictions).

A database was created of all relevant publications found during searches. Reasons for exclusion were recorded for all those included during screening that were not summarized for the synopsis (see section 1.6.2).

1.6.2 Publication screening and inclusion criteria

A summary of the total number of evidence sources screened is presented in Appendix 3. The initial screening process is at the title and abstract level. If selected following this initial screening, a second one at the full-text level is undertaken, to validate whether the study fits the Conservation Evidence inclusion criteria (described below).

a) Screening

To ensure consistency/accuracy when screening publications for inclusion in the literature database, an initial test using the Conservation Evidence inclusion criteria (provided below) and a consistent set of references was carried out by the authors, compared with the decisions of the experienced core Conservation Evidence team. Results were analysed using Cohen's Kappa test (Cohen 1960). Where initial results did not show 'substantial' (K = 0.61–0.8) or 'almost perfect' agreement (K = 0.81–1.0), authors were given further training. A second Kappa test was used to assess the consistency/accuracy of article screening for the first two years of the first journal searched by each author. Again, where results did not show 'substantial' (K = 0.61–0.8) or 'almost perfect' agreement (K = 0.81–1.0), authors received further training and were tested again before carrying out further searches.

We acknowledge that the literature search and screening method used, as with any method, results in gaps in the evidence. Keyword search terms are not exhaustive and searchers may have missed relevant papers during screening. Potential publication bias is not taken into account, and it is likely that additional biases will result from the evidence that is available. For example, there are often geographic biases in study locations and bias towards publication of significant results, with failures and/or negative outcomes more likely to go unreported (Firth *et al.* 2020).

b) Inclusion criteria

The following Conservation Evidence inclusion criteria were used.

Criteria A: Conservation Evidence includes studies that measure the effect of an action that might be done to conserve biodiversity

1. Does this study measure the effect of an action that is or was under the control of humans, on wild taxa (including captives), habitats, or invasive/problem taxa? If yes, go to 3. If no, go to 2.
2. Does this study measure the effect of an action that is or was under the control of humans, on human behaviour that is relevant to conserving biodiversity? If yes, go to Criteria B. If no, exclude.

3. Could the action be put in place by a conservationist/decision-maker to protect, manage, restore or reduce impacts of threats to wild taxa or habitats, or control or mitigate the impact of the invasive/problem taxon on wild taxa or habitats? If yes, include. If no, exclude.

Explanation:

1. a. Study must have a measured outcome on wild taxa, habitats or invasive species: excludes studies on domestic/agricultural species, theoretical modelling or opinion pieces. See Criteria B for actions that have a measured outcome on human behaviour only.

1. b. Action must be carried out by people: excludes impacts from natural processes (e.g. storm events, erosion), impacts from background variation (e.g. salinity, wave exposure, climate change), correlations with habitat types, where there is no test of a specific action by humans, or pure ecology (e.g. movement, distribution of species).

2. Study must test an action that could be put in place for conservation. This excludes assessing impacts of threats (actions which remove threats would be included). The test may involve comparisons between sites/factors not originally put in place or modified for conservation, but which could be (e.g. seawalls with and without species transplanted onto them for commercial culturing purposes, maintained structures vs structures where maintenance stopped – where the transplantation/maintenance cessation is as you would do for conservation, even if that was not the original intention in the study).

If the title and/or abstract are indicative of fulfilling our criteria, but you do not have sufficient information to judge whether the action was under human control, the action could be applied by a conservationist/decision-maker or whether there are data quantifying the outcome, then include. If the article has no abstract, but the title is indicative that it might test a relevant intervention, then include. It is possible that some relevant publications are missed at this stage if the title is not deemed indicative by the author undertaking the search.

We sort articles into folders by which taxon/habitat they report an outcome on. If the title/abstract does not specify which species/taxa/habitats are impacted, then please scan the full article and then assign to folders accordingly.

The outcome for wild taxa/habitats can be negative, neutral or positive, does not have to be statistically significant but must be quantified (if hard to judge from abstract, then include). It could be any outcome that has implications for the health of individuals, populations, species, communities or habitats, including, but not limited to the following:

- Individual health, condition or behaviour, including in captivity: growth, size, weight, stress, disease levels or immune function, movement, use of natural/artificial habitat/structure, range, or predatory or nuisance behaviour that could lead to retaliatory action by humans

- Breeding: egg/larvae/sperm production, mating success, birth rate, clutch size, 'overall recruitment'
- Genetics: genetic diversity, genetic suitability (e.g. adaptation to local conditions)
- Life history: age/size at maturity, survival, mortality
- Population measures: number, abundance, density, presence/absence, biomass, movement, cover, age-structure, species distributions (only in response to a human action), disease prevalence, sex ratio
- Community/habitat measures: species richness, diversity measures (including trait/functional diversity), community composition, community structure (e.g. trophic structure), area covered, physical habitat structure (e.g. rugosity, height, basal area)

Actions within the scope of Conservation Evidence include:

- Clear management actions: creating artificial habitats or shelters, transplanting or seeding vegetation, ceasing or altering damaging maintenance activities
- International, national, or local policies: creating marine protected areas, bylaws, local voluntary restrictions
- Reintroductions of wild species in captivity
- Actions that reduce human-wildlife conflict
- Actions that change human behaviour, resulting in an impact on wild taxa or habitats

See https://www.conservationevidence.com/data/index for more examples of actions

Note on study types:

Include any literature reviews, systematic reviews, meta-analyses or short notes that review studies that fulfil these criteria.

Exclude theoretical modelling studies, as no action has been taken. However, studies that use models to analyse real-world data, or compare models to real-world situations are included (if they otherwise fulfil these criteria).

Criteria B: Conservation Evidence includes studies that measure the effect of an action that might be done to change human behaviour for the benefit of biodiversity

1. Does this study measure the effect of an action that is or was under human control on human behaviour (actual or intentional) which is likely to protect, manage, restore or reduce threats to wild taxa or habitats? If yes, go to 2. If no, exclude.

2. Could the action be put in place by a conservationist, manager or decision-maker to change human behaviour? If yes, include. If no, exclude.

Explanation:

1. a. Study must have a measured outcome on actual or intentional human behaviour including self-reported behaviours: excludes outcomes on human psychology (tolerance, knowledge, awareness, attitude, perceptions or beliefs)

1. b. Change in human behaviour must be linked to outcomes for wild taxa and habitats, excludes changes in behaviour linked to outcomes for human benefit, even if these occurred under a conservation programme (e.g. we would exclude a study demonstrating increased school attendance in villages under a community-based conservation programme)

1. c. Action must be under human control: excludes impacts from climatic or other natural events.

2. Study must test an action that could be put in place for conservation: excludes studies with no action e.g. correlating human personality traits with likelihood of conservation-related behaviours.

The human behaviour outcome of the study can be negative, neutral or positive, does not have to be statistically significant but must be quantified (if hard to judge from abstract, then include). It could be any behaviour that is likely to have an outcome on wild taxa and habitats (including mitigating the impact of invasive/problem taxon on wild taxa or habitats). Actions include, but are not limited to the following:

- Change in adverse behaviours (which directly threaten biodiversity): e.g. unsustainable or illegal fishing, urban encroachment, creating noise, entering sensitive areas, polluting or dumping waste, clearing or habitat destruction, introducing invasive species.
- Change in positive behaviours: e.g. uptake of alternative/sustainable livelihoods, number of households adopting sustainable practices, donations
- Change in policy or conservation methods: e.g. placement of protected areas, protection of key habitats/species
- Change in consumer or market behaviour: e.g. purchasing, consuming, buying, willingness to pay, selling, illegal trading, advertising, consumer fraud.

Actions that are particularly likely to induce a human behaviour change include, but are not limited to the following:

- Enforcement: closed seasons, size limits, fishing/hunting gear restrictions, auditable/traceable reporting requirements, market inspections, increase number of rangers, patrols or frequency of patrols in, around or within protected areas, improve fencing/physical barriers, improve signage, improve equipment/technology used by guards, use of Unmanned Autonomous Vehicles/drones for rapid response, DNA analysis, GPS tracking.
- Behaviour Change: promote alternative/sustainable livelihoods, payment for ecosystem services, ecotourism, poverty reduction, debunking misinformation, altering or re-enforcing local taboos, financial incentives.
- Governance: protect or reward whistle-blowers, increase government transparency, ensure independence of judiciary, provide legal aid.
- Market Regulation: trade bans, taxation, supply chain transparency laws.
- Consumer Demand Reduction: fear appeals (negative association with undesirable product), benefit appeal (positive association with desirable behaviour), worldview framing, moral framing, employing decision defaults, providing decision support tools, simplifying advice to consumers, promoting desirable social norms, legislative prohibition.
- Sustainable Alternatives: certification schemes, captive bred or artificial alternatives, sustainable alternatives.
- New policies and regulations for conservation/protection: hard laws, soft laws, voluntary regulations.

We allocate studies to folders by their outcome. All studies under Criteria B go in the 'Human behaviour change' folder. They are additionally duplicated into a taxon/habitat folder if there is a specific intended final outcome of the behaviour change (if none mentioned, file only in Human behaviour change).

c) Relevant subject

Studies relevant to the synopsis subject include those focussed on enhancing the wild, native biodiversity of marine artificial structures carried out in intertidal and subtidal marine habitats, including in estuaries, along coastlines and offshore.

d) Relevant types of intervention

An intervention has to be one that could be put in place by a developer or landowner whose development proposal includes building or modifying marine artificial structures, a marine planning advisor, an engineer or designer, a conservationist, a community group, a marine protected area manager, or a policy maker, to enhance the wild native biodiversity of marine artificial structures, or to control or mitigate the impact of an invasive/problem taxon. Alternatively, interventions may aim to change human behaviour (actual or intended), which is likely to enhance the biodiversity of marine artificial structures. See inclusion criteria above for further details.

If the following two criteria were met, a combined intervention was created within the synopsis, rather than repeating evidence under all the separate interventions: a) there are five or more publications that use the same well-defined combination of interventions, with very clear descriptions of what they were, without separating the effects of each individual intervention, and b) the combined set of interventions is a commonly-used conservation strategy.

e) Relevant types of comparator

To determine the effectiveness of interventions, studies should include a comparison, for example, monitoring change over time (typically before and after the intervention was implemented), or comparing 'treatment' sites where an intervention was undertaken and 'control' sites where no intervention took place. Alternatively, a study could compare one specific intervention (or implementation method) against another. For example, a study could compare the abundance of a species colonizing seawalls before and after 'rock pool' habitats were created on them, or its abundance in two different created 'rock pool' designs. Acknowledging that there are various barriers to experimenting on entire full-scale marine artificial structures (e.g. permissions, costs, risks to the environment or engineering integrity), patch-scale comparisons are also included. For example, a study could compare the abundance of a species colonizing 'rock pools' created on a seawall with its abundance on seawall surfaces of comparable patch-size without created pools, or a study could compare environmentally-sensitive materials with standard construction materials using settlement plates (patches) of different materials attached to a seawall.

Exceptions lacking one of the suitable comparators listed above may still be included, for example, where a comparator is not essential to at least partially assess the effectiveness of the intervention (e.g. uptake of created habitats or shelters by target species; survival of organisms transplanted onto structures).

f) Relevant types of outcome

Below we provide a list of anticipated metrics; others are included if reported within relevant studies.

- Community response:
  - Community composition (also used to describe assemblage compositions)
  - Richness/diversity
- Population response:
  - Abundance: number, density, presence/absence, biomass
  - Reproductive success: egg/larvae production, mating success, hatching rate, egg/larvae quality/condition
  - Survival: survival, mortality
  - Condition: growth, size, weight, condition factors (condition indices), biochemical ratios, stress, disease levels
- Behaviour:
  - Use by species of created habitats or shelters
  - Species behaviour change: movement patterns, feeding activity
  - Human behaviour change

g) Relevant types of study design

Table 2 lists the study designs included. The strongest evidence comes from replicated, randomized, controlled trials with paired-sites and before-and-after monitoring. For further information on study designs and their quality or strength, please see Christie *et al.* (2019).

Table 2. Study designs

Term	**Meaning**
Replicated	The intervention was repeated on more than one individual or site. In conservation and ecology, the number of replicates is much smaller than it would be for medical trials (when thousands of individuals are often tested). If the replicates are sites, pragmatism dictates that between five and ten replicates is a reasonable amount of replication, although more would be preferable. We provide the number of replicates wherever possible. Replicates should reflect the number of times an intervention has been independently carried out, from the perspective of the study subject. For example, 10 patches on a seawall might be independent replicates from the perspective of algae and non-mobile invertebrates with limited dispersal, but not for larger motile animals such as fishes.

Randomized	The intervention was allocated randomly to individuals or sites. This means that the initial condition of those given the intervention is less likely to bias the outcome.
Paired sites	Sites are considered in pairs, within which one was treated with the intervention and the other was not. Pairs, or blocks, of sites are selected with similar environmental conditions, such as surface aspect/orientation or surrounding seascape. This approach aims to reduce environmental variation and make it easier to detect a true effect of the intervention.
Controlled*	Individuals or sites treated with the intervention are compared with control individuals or sites not treated with the intervention. (The treatment is usually allocated by the investigators (randomly or not), such that the treatment or control groups/sites could have received the treatment).
Before-and-after	Monitoring of effects was carried out before and after the intervention was imposed.
Site comparison*	A study that considers the effects of interventions by comparing sites that historically had different interventions (e.g. intervention vs no intervention) or levels of intervention. Unlike controlled studies, it is not clear how the interventions were allocated to sites (i.e. the investigators did not allocate the treatment to some of the sites).
Review	A conventional review of literature. Generally, these have not used an agreed search protocol or quantitative assessments of the evidence.
Systematic review	A systematic review follows structured, predefined methods to comprehensively collate and synthesize existing evidence. It must weight or evaluate studies, in some way, according to the strength of evidence they offer (e.g. sample size and rigour of design). Environmental systematic reviews are available at https://environmentalevidence.org/.

Study	If none of the above apply, for example a study measuring change over time in only one site and only after an intervention. Or a study measuring use of created habitats at one site.

* Note that 'controlled' is mutually exclusive from 'site comparison'. A comparison cannot be both controlled and a site comparison. However, one study might contain both controlled and site comparison aspects, e.g. study of 'rock pool' habitats created on seawalls, compared with seawall surfaces without created pools (controlled) and natural rock pools on a natural rocky reef (site comparison).

1.6.3 Study quality assessment & critical appraisal

We did not quantitatively assess the evidence from each publication or weight it according to quality. However, to allow interpretation of the evidence, we made the size, design and duration of each study we reported clear.

We critically appraised each potentially relevant study and excluded those that did not provide data for a comparison to the treatment (but see exceptions above), did not statistically analyse the results (or if included this was stated in the summary paragraph), or had obvious errors in their design or analysis. A record of the reason for excluding any of the publications included during screening was kept within the synopsis database.

1.6.4 Data extraction

Data on the performance/effect of the relevant intervention (e.g. average species richness inside vs outside created habitats; survival of organisms transplanted onto structures) were extracted from, and summarized for, publications that included the relevant subject, types of intervention, comparator and outcomes outlined above.

At the start of each month, authors swapped three summaries with another author to ensure that the correct type of data had been extracted and that the summary followed the Conservation Evidence standard format.

1.6.5 Evidence synthesis

a) Summary protocol

Each publication usually had just one paragraph for each intervention it tested, describing the study in (usually) no more than 200 words using plain English. To help with some of the terminology specific to the marine environment, and for which plain English equivalents do not exist, we provide a Glossary of terms (Appendix 1). Each summary used the following format:

A [TYPE OF STUDY] in [YEARS X-Y] in [HOW MANY SITES] in/of [STRUCTURE/HABITAT TYPE] in [ENVIRONMENTAL CONTEXT, WATER BODY and COUNTRY] [REFERENCE] found that [INTERVENTION] [SUMMARY OF ALL KEY RESULTS] for [SPECIES/HABITAT TYPE]. [DETAILS OF KEY RESULTS, INCLUDING DATA]. In addition, [EXTRA RESULTS, IMPLEMENTATION OPTIONS, CONFLICTING RESULTS]. The [DETAILS OF EXPERIMENTAL DESIGN, INTERVENTION METHODS and KEY DETAILS OF SITE CONTEXT]. Data was collected in [DETAILS OF SAMPLING METHODS].

Type of study – see terms and order in Table 2.

Results – publications often contain more results than can be summarized effectively in 200 words. Only key results relevant to the effects of the intervention are included. Where interventions were monitored over time, final survey data are reported for non-mobile species/communities that are expected to follow successional trajectories. For highly mobile species, results over all repeated surveys are reported where possible. Readers are referred to the original source if there are additional or more detailed results for individual species that are not included within the summary.

Methods – for the sake of brevity, only nuances essential to the interpretation of the results are included. The reader is always encouraged to read the original source to get a full understanding of the study sites (e.g. physical conditions, history of management) and survey methods (e.g. details of sampling regime, specific equipment used).

For example:

A replicated, paired sites, controlled study in 2014–2015 on one intertidal breakwater and two groynes on open coastline in the Alboran Sea, Spain (1) found that rock pools created on the structures supported different macroalgae and invertebrate community composition with higher species diversity and richness than structure surfaces without pools. After 12 months, macroalgae and invertebrate species diversity (data reported as Shannon index) and richness were higher in pools (9 species/pool) than on structure surfaces without (6/surface), and the community composition differed (data reported as statistical model results). Upper-midshore pools supported similar richness to lower-midshore ones (8 vs 9 species/pool). Eight species (4 macroalgae, 4 mobile invertebrates) recorded in pools were absent from structure surfaces without. Rock pools were created in February 2014 by drilling into horizontal surfaces of three limestone boulder structures (1 breakwater, 2 groynes; treated as 1 site) using a jackhammer. Five irregularly-shaped pools (average length: 176 mm; width: 137 mm; depth: ≤20 mm; volume: 0.4 l) were drilled at both upper-midshore and lower-midshore on each structure. Pools were compared with breakwater/groyne surfaces (200 × 200 mm) adjacent to each pool. Macroalgae and invertebrates were counted in pools and on structure surfaces from photographs over 12 months.

(1) Ostalé-Valriberas E., Sempere-Valverde J., Coppa S., García-Gómez J.C. & Espinosa F. (2018) Creation of microhabitats (tidepools) in ripraps with climax communities as a way to mitigate negative effects of artificial substrate on marine biodiversity. *Ecological Engineering*, 120, 522–531.

A replicated, randomized, controlled study in 2005 on three subtidal rocky reefs on open coastlines in the Adriatic Sea and the Ionian Sea, Italy (2) found that settlement plates with and without textured surfaces supported similar macroalgae and non-mobile invertebrate species richness, live cover and community composition, while abundances varied depending on the species group and site. After nine months, there was no clear difference in the macroalgae and non-mobile invertebrate community composition, species richness or live cover on plates with and without textured surfaces (data reported as statistical model results). Non-mobile invertebrates were more abundant on plates with texture (<1–6% cover) than without (<1–2%) but the difference was only significant at one of six sites. Macroalgal abundances varied by species group and site (see paper for results). Limestone, sandstone, granite and concrete settlement plates (150 × 100 mm) were made with and without textured surfaces. Five of each material-texture combination were randomly arranged horizontally at 5 m depth in each of two sites on each of three limestone rocky reefs in February 2005. Macroalgae and non-mobile invertebrates on plates were counted in the laboratory over nine months.

(2) Guarnieri G., Terlizzi A., Bevilacqua S. & Fraschetti S. (2009) Local vs regional effects of substratum on early colonization stages of sessile assemblages. *Biofouling: The Journal of Bioadhesion and Biofilm Research*, 25, 593–604.

b) Terminology used to describe the evidence

Unless specifically stated otherwise, results reflect statistical tests performed on the data, i.e. we only state that there was a difference if it was a statistically significant difference or state that there was no difference if it was not significant. Table 2 above defines the terms used to describe the study designs.

c) Dealing with multiple interventions within a publication

When separate results are provided for the effects of each of the different interventions tested, separate summaries have been written under each intervention heading. However, when several interventions were carried out at the same time and only the combined effect reported, the results were described with a similar paragraph under all relevant interventions. The first sentence makes it clear that there was a combination of interventions carried out, i.e. '...(REF) found that [x intervention], along with [y] and [z interventions] resulted in [describe effects]'. Within the results section we also added a sentence: 'It is not clear whether these effects were a direct result of [x], [y] or [z] interventions'.

d) Dealing with multiple publications reporting the same results

If two publications described results from the same intervention implemented in the same space and at the same time, we only included the most stringently peer-reviewed publication (i.e. if a study is published in an academic journal and in a book chapter, we would include the academic journal). If one included initial results (e.g. after year one) of another (e.g. after 1–3 years), we only included the publication covering the longest time span. If two publications described at least partially different results, we included both but made clear they were from the same project in the paragraph, e.g. 'A controlled study... (Gallagher *et al.* 1999; same experimental set-up as Oasis *et al.* 2001)...'.

e) Taxonomy

Taxonomy was not updated but followed that used in the original publication. Where possible, common names and Latin/scientific names were both given the first time each species was mentioned within each summary.

f) Key messages

Each intervention has a set of concise, bulleted key messages at the top, which was written once all the literature had been summarized. These include information such as the number, design and location of studies included.

The first bullet point describes the total number of studies that tested the intervention and the locations of the studies, followed by key information on the relevant metrics presented under the headings and sub-headings shown below (with number of relevant studies in parentheses for each).

- **X studies** examined the effects of [INTERVENTION] on [TARGET POPULATION]. Y studies were in [LOCATION 1][1,2] and Z studies were in [LOCATION 2][3,4].
  Here, locations include a description of the broad environmental context (e.g. on open coastlines, in estuaries, etc.) and country, ordered based on descending number of studies in each type of context, then chronologically, rather than alphabetically, i.e. USA[1], Australia[2], not Australia[2], USA[1].

COMMUNITY RESPONSE (x STUDIES)

- **Community composition* (x studies):**
- **Richness/diversity (x studies):**

POPULATION RESPONSE (x STUDIES)

- **Abundance (x studies):**
- **Reproductive success (x studies):**
- **Survival (x studies):**
- **Condition (x studies):**

BEHAVIOUR (x STUDIES)

- **Use (x studies):**
- **Behaviour change (x studies):**

OTHER (x STUDIES) *(Included only for interventions/chapters where relevant)*

- **[Sub-heading(s) for the metric(s) reported will be created as necessary] (x studies):**

*Also used to refer to assemblage composition

If no evidence was found for an intervention, the following text was added in place of the key messages above:

- We found no studies that evaluated the effects of [INTERVENTION] on [TARGET POPULATION].

  This means we did not find any studies that directly evaluated this intervention during our literature searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

g) Background information

Background information for each intervention is provided, with relevant references, directly after the key messages, before the individual evidence summaries are presented. Here we describe and define the intervention and outline its ecological rationale. We also highlight wider knowledge about the intervention and potential associated risks to help the reader interpret the evidence. In some cases, where a body of literature has strong implications for enhancing the biodiversity of marine artificial structures, but does not directly test the effects of an intervention, we refer the reader to this literature. We also direct the reader to other related or similar interventions that may be of interest.

1.6.6 Dissemination/communication of evidence synthesis

The information from this synopsis will be available in three ways:

- This synopsis pdf, downloadable from www.conservationevidence.com, which contains the study summaries, key messages and background information on each intervention.
- The searchable database at www.conservationevidence.com, which contains all the summarized information from the synopsis, along with expert assessment scores.
- A chapter in *What Works in Conservation*, available as a pdf to download and a book from https://www.conservationevidence.com/content/page/79, which contains the key messages from the synopsis as well as expert assessment scores on the effectiveness and certainty of the synopsis, with links to the online database.

1.7. How to use the information provided

This synopsis can be used to guide conservation actions and management plans. It can be referred to during the planning, design, consenting, operational and/or decommissioning phases of marine artificial structures. However, it does not tell you what to do.

To use this synopsis effectively, we recommend that you search for information relevant to your work (see Appendix 2 for intervention list with definitions and scales), and then assess how applicable the interventions are to your situation. For example, ask yourself:

- Do they deal with the same types of structures in the same environmental contexts?
- Do they report outcomes for the same types of target species/communities/habitats?
- Which studies are the most relevant?
- How dependent were the outcomes on local conditions?
- For how long and at what scale were the effects monitored?
- What comparators were used to measure effects?
- How strong is the evidence one way or another?
- What are the wider environmental risks and carbon footprint of implementing the interventions?

Apply the information to your situation and decide on the course of action most likely to deliver your desired outcomes. We suggest that you refer to the original source to gain a full understanding of particular studies.

IMPORTANT NOTE – Interpreting the evidence

Care must be taken when interpreting some of the evidence provided. Studies do not always measure the most appropriate metric or assess effects at the population or the full artificial structure level. For example, a small proportion of crabs using 'rock pool' habitats created on a seawall does not make it an effective intervention if a greater proportion are using adjacent (unsurveyed) seawall surfaces or other habitats without created pools. Furthermore, if no measure of those crabs' condition, behaviour or survival are reported then it is not possible to know if those created habitats are providing beneficial refuges or ecological 'traps', potentially with a negative overall impact on local crab populations.

The duration over which effects have been evaluated must also be considered, given that community development can take several years and effects on populations may require long-term monitoring to be detected. In addition, it should be recognized that the timing (e.g. season) of an intervention is likely to affect what species settle onto surfaces and occupy space first. This will depend on what larvae/spores are present in the plankton at the time, and can have knock-on effects on later arrivals and ultimate community development. This is particularly important in locations where non-native invasive species are present. Other factors that are likely to influence biodiversity outcomes include, but are not limited to, the environmental context of the structures (e.g. latitude, water quality, salinity, wave exposure, flow dynamics, aspect, shading, sound), disturbance regimes (e.g. human activity, sand scour/burial, pollution), position in the water column (intertidal shore level, subtidal depth), and proximity of other habitats dictating the available species pool in the area.

Finally, a lack of evidence does not mean that interventions are not effective for enhancing the biodiversity of marine artificial structures, or that such measures should be abandoned. It simply highlights the need for further research and robust monitoring in these areas to ensure that future conservation efforts will be appropriate and effective.

1.8. How you can help to change conservation practice

If you know of evidence relating to enhancing the biodiversity of marine artificial structures that is not included in this synopsis, we invite you to contact us via our website www.conservationevidence.com. If you have new, unpublished evidence, you can submit a paper to the Conservation Evidence journal. We particularly welcome papers submitted by conservation practitioners.

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2. Enhancing the biodiversity of intertidal artificial structures

Background

Intertidal artificial structures (or portions of structures) are covered and uncovered by seawater during daily tidal cycles. They include, but are not limited to, seawalls, breakwaters, groynes and jetties commonly found along coastlines, and also intertidal portions of structures in tidal waters further offshore, such as wind turbine pilings. The names given to different types of structures vary with geography, so we provide a Glossary of terms in Appendix 1 with alternative names for those used here.

Intertidal artificial structures tend to support lower biodiversity than natural intertidal rocky habitats and are often colonized by weedy or opportunistic species, including non-natives (Airoldi *et al.* 2015). There are various reasons for their reduced biodiversity. Structures often have steeper inclinations than natural reefs, with narrower bands of intertidal habitat, meaning that space for organisms is scarce and competitive interactions and other environmental processes differ (Chapman & Underwood 2011). The materials used in construction can be unfavourable for certain species (Dennis *et al.* 2018), while the uniform shapes and flat surfaces of many structures offer low habitat complexity with fewer niche spaces (Lawrence *et al.* 2021; Moschella *et al.* 2005). Furthermore, structure designs can present novel habitat conditions not otherwise found in nature (e.g. sheltered sides of shore-parallel breakwaters built on exposed coastlines; shaded surfaces on pilings under jetties).

The eco-engineering approaches described here focus on altering structure designs to increase their habitat complexity and niche availability, with the aim of enhancing their biodiversity. They also explore using alternative construction materials, transplanting or seeding species directly onto structures, controlling or removing non-native or nuisance species, and managing human disturbances from maintenance and harvesting activities. It is important to recognize the importance of the environmental context in which structures are placed in influencing the biodiversity that can colonize and survive on them, and thus the likely effects of interventions in different scenarios. Structures built in urban environments may be subject to high human disturbance (Airoldi & Bulleri 2011) and poor water quality (Perrett *et al.* 2006). Those built on exposed sandy or muddy shorelines (i.e where coastal protection or reinforcement is often required) may be frequently disturbed by wave energy and sediment scouring/burial (Moschella *et al.* 2005), and may be too far from source populations for some species to colonize.

It is crucial, therefore, that decision-makers understand the ecology of the species and communities they wish to target with their actions to enhance biodiversity on intertidal artificial structures, and the environmental context in which their structures are located. We encourage the reader to take particular note of the location and context in which studies summarized here were carried out, along with the spatial scale and timeframe over which effects were monitored.

This chapter describes 22 conservation interventions that could be carried out to enhance the biodiversity of intertidal artificial structures or intertidal portions of artificial structures that also extend into the subtidal. We found evidence for the effects of 18 of these interventions. Definitions are provided in the background sections for each intervention (also see intervention list in Appendix 2). These are particularly important for interventions that involve creating artificial habitats or shelters. There has been little consistency in the literature to date in naming conventions for habitats occurring in nature or created for conservation intervention. One person's 'crevice' may be another's 'groove'. Here, we define habitats according to their size and shape. The "See also" sections at the end of each background signpost the reader to similar or related interventions.

Airoldi L. & Bulleri F. (2011) Anthropogenic disturbance can determine the magnitude of opportunistic species responses on marine urban infrastructures. *PLoS ONE*, 6, e22985.

Airoldi L., Turon X., Perkol-Finkel S. & Rius M. (2015) Corridors for aliens but not for natives: effects of marine urban sprawl at a regional scale. *Diversity and Distributions*, 21, 755–768.

Chapman M.G. & Underwood A.J. (2011) Evaluation of ecological engineering of "armoured" shorelines to improve their value as habitat. *Journal of Experimental Marine Biology and Ecology*, 400, 302–313.

Dennis H.D., Evans A.J., Banner A.J. & Moore P.J. (2018) Reefcrete: reducing the environmental footprint of concretes for eco-engineering marine structures. *Ecological Engineering*, 120, 668–678.

Lawrence P.J., Evans A.J., Jackson-Bué T., Brooks P.R., Crowe T.P., Dozier A.E., Jenkins S.R., Moore P.J., Williams G.J. & Davies A.J. (2021) Artificial shorelines lack natural structural complexity across scales. *Proceedings of the Royal Society B*, 288, 20210329.

Moschella P.S., Abbiati M., Åberg P., Airoldi L., Anderson J.M., Bacchiocchi F., Bulleri F., Dinesen G.E., Frost M., Gacia E., Granhag L., Jonsson P.R., Satta M.P., Sundelöf A., Thompson R.C. & Hawkins S.J. (2005) Low-crested coastal defence structures as artificial habitats for marine life: using ecological criteria in design. *Coastal Engineering*, 52, 1053–1071.

Perrett L.A., Johnston E.L. & Poore A.G.B. (2006) Impact by association: direct and indirect effects of copper exposure on mobile invertebrate fauna. *Marine Ecology Progress Series*, 326, 195–205.

2.1. Use environmentally-sensitive material on intertidal artificial structures

- **Eight studies** examined the effects of using environmentally-sensitive material on intertidal artificial structures on the biodiversity of those structures. Three studies were on open coastlines in the UK[1,3] and Ireland[8], and one was in each of an estuary in southeast Australia[2], a marina in northern Israel[4], and a port in southeast Spain[5]. One was on an open coastline and in estuaries in the UK[6], and one was on island coastlines in the Singapore Strait and in estuaries in the UK[7].

COMMUNITY RESPONSE (6 STUDIES)

- **Overall community composition (4 studies):** Two of four replicated, controlled studies (including three randomized and one paired sites, before-and-after study) in Australia[2], the UK[3], Israel[4], and Singapore and the UK[7], found that using hemp-concrete in place of standard-concrete[3] on intertidal artificial structures, or using ECOncrete™, along with creating grooves, small ledges and holes[4], altered the combined macroalgae and invertebrate community composition on structure surfaces. One of the studies[3], along with one other[7], found that using shell-concrete[3] or reduced-pH-concrete[7] did not. One study[2] found that using sandstone in place of basalt had mixed effects, depending on the site. Two of the studies[4,7] reported that ECOncrete™ surfaces with added habitats[4] or reduced-pH-concrete surfaces[7] supported macroalgae[4], mobile invertebrate[7] and/or non-mobile invertebrate[4,7] species that were absent from standard-concrete structure surfaces.

- **Algal community composition (1 study):** One replicated, randomized, paired sites, controlled study in Spain[5] found that using different materials (sandstone, limestone, slate, gabbro, concrete) on an intertidal artificial structure altered the diatom community composition on structure surfaces.
- **Overall richness/diversity (4 studies):** Two of four replicated, controlled studies (including three randomized and one paired sites, before-and-after study) in the UK[3,6], Israel[4], and Singapore and the UK[7] found that using hemp-concrete[3], shell-concrete[3] or reduced-pH-concrete[7] in place of standard-concrete on intertidal artificial structures did not increase the combined macroalgae and invertebrate species richness on structure surfaces. One study[4] found that using ECOncrete™, along with creating grooves, small ledges and holes, did increase the species richness and diversity. One[6] found that using limestone-cement, along with creating pits, grooves, small ridges and texture, had mixed effects depending on the site.
- **Algal richness/diversity (1 study):** One replicated, randomized, paired sites, controlled study in Spain[5] found that using quarried rock in place of concrete on an intertidal artificial structure did not increase the diatom species richness or diversity on structure surfaces.
- **Invertebrate richness/diversity (1 study):** One replicated, randomized, controlled study in the UK[3] found that using hemp-concrete in place of standard-concrete on intertidal artificial structures increased the mobile invertebrate species richness on structure surfaces, but using shell-concrete did not.

POPULATION RESPONSE (7 STUDIES)

- **Overall abundance (1 study):** One replicated, randomized, controlled study in the UK[3] found that using hemp-concrete or shell-concrete in place of standard-concrete on intertidal artificial structures increased the combined macroalgae and non-mobile invertebrate abundance on structure surfaces.
- **Algal abundance (5 studies):** Four of five replicated, controlled studies (including four randomized and one paired sites study) in Australia[2], Spain[5], Singapore[7], the UK[1,7] and Ireland[8] found that using sandstone in place of basalt[2], quarried rock in place of concrete[1,5], or altering the composition of concrete[8] on intertidal artificial structures had mixed effects on the macroalgal[2] or microalgal[1,5,8] abundance on structure surfaces, depending on the species group[1,2,8], site[2], wave-exposure[8] and/or the type of material tested[1,5,8]. One study[7] found no effect of reducing the pH of concrete on macroalgal abundance.
- **Invertebrate abundance (4 studies):** Two of four replicated, controlled studies (including three randomized studies) in Australia[2], the UK[6], Singapore and the UK[7] and Ireland[8] found that using sandstone in place of basalt[2] or reducing the pH of concrete[7] on intertidal artificial structures did not increase the abundance of tubeworms[2], oysters[2], limpets[7], barnacles[2,7] and/or combined invertebrates[7] on structure surfaces. Two studies[6,8] found that using limestone-cement[6], along with creating pits, grooves, small ridges and texture, or altering the composition of concrete[8] had

mixed effects on the mobile invertebrate[6] and/or barnacle[6,8] abundance, depending on the site[6], wave-exposure[8] and/or the type of material tested[8].

BEHAVIOUR (0 STUDIES)

Background

Material type influences the settlement and survival of marine organisms in intertidal rocky habitats. Settlement preferences and competition lead to some species being more abundant than others on certain materials (Green *et al.* 2012; Iveša *et al.* 2010), but patterns vary by environmental conditions. Physical (lithology, hardness, porosity, colour, texture) and chemical (pH, mineralogy, toxicity) properties of rock and manufactured materials can affect how they weather over time and what communities develop on them (Coombes *et al.* 2011).

Marine artificial structures are often made from hard quarried rock, concrete, wood, steel or plastic, according to engineering requirements, cost and/or availability. Synthetic materials can be associated with the presence of non-native species (Dafforn 2017), whereas structures made from natural rock may support more natural rocky reef communities. There may be opportunities to use more environmentally-sensitive materials in structures or in eco-engineering habitat designs added to structures to enhance their biodiversity. Concrete is commonly-used in eco-engineering since it is durable and easy to mould into complex shapes. Yet adding manufactured concrete habitats to structures to enhance biodiversity may not deliver a net environmental gain because of the large CO_2 footprint of concrete production (Heery *et al.* 2020). Concrete mixes can be manipulated to alter their physical and chemical properties (McManus *et al.* 2018; Natanzi *et al.* 2021) and environmental footprint (Dennis *et al.* 2018). Lower-footprint materials may be preferable, regardless of their effect on colonizing biodiversity; a neutral/no effect on biodiversity may still offer a higher net environmental gain (or lower net loss).

It is often not possible to separate the effects of the various physical and chemical properties of materials on biodiversity. Studies that directly examine the effects of creating different surface textures are included under the action "*Create textured surfaces (≤1 mm) on intertidal artificial structures*"; any other material comparisons are considered here. There are bodies of literature investigating the effects of material on settlement behaviour and ecological interactions in the laboratory and field (e.g. Anderson 1996; Iveša *et al.* 2010; Herbert & Hawkins 2006) and for anti-fouling applications (e.g. Hanson & Bell 1976). These studies are not included in this synopsis, which focusses on *in situ* conservation actions to promote colonization of biodiversity on marine artificial structures.

Definition: 'Environmentally-sensitive materials' are materials that seek to maximize environmental benefits and/or minimize environmental risks of marine engineering.

See also: *Create textured surfaces (≤1 mm) on intertidal artificial structures.*

Anderson M.J. (1996) A chemical cue induces settlement of Sydney rock oysters, *Saccostrea commercialis*, in the laboratory and in the field. *The Biological Bulletin*, 190, 350–358.
Coombes M.A., Naylor L.A., Thompson R.C., Roast S.D., Gómez-Pujol L. & Fairhurst R.J. (2011) Colonization and weathering of engineering materials by marine microorganisms: an SEM study. *Earth Surface Processes and Lanforms*, 36, 585–593.
Dafforn K.A. (2017) Eco-engineering and management strategies for marine infrastructures to reduce establishment and dispersal of non-indigenous species. *Management of Biological Invasions*, 8, 153–161.
Dennis H.D., Evans A.J., Banner A.J. & Moore P.J. (2018) Reefcrete: reducing the environmental footprint of concretes for eco-engineering marine structures. *Ecological Engineering*, 120, 668–678.
Green D.S., Chapman M.G. & Blockley D.J. (2012) Ecological consequences of the type of rock used in the construction of artificial boulder-fields. *Ecological Engineering*, 46, 1–10.
Hanson C.H. & Bell J. (1976) Subtidal and intertidal marine fouling on artificial substrata in northern Puget Sound, Washington. *Fishery Bulletin* (United States), 74, 2.
Heery E.C., Lian K.Y., Loke L.H.L., Tan H.T.W. & Todd P.A. (2020) Evaluating seaweed farming as an eco-engineering strategy for 'blue' shoreline infrastructure. *Ecological Engineering*, 152, 105857.
Herbert R.J.H. & Hawkins S.J. (2006) Effect of rock type on the recruitment and early mortality of the barnacle *Chthamalus montagui*. *Journal of Experimental Marine Biology and Ecology*, 334, 96–108.
Iveša L., Chapman M.G., Underwood A.J. & Murphy R.J. (2010) Differential patterns of distribution of limpets on intertidal seawalls: experimental investigation of the roles of recruitment, survival and competition. *Marine Ecology Progress Series*, 407, 55–69.
McManus R.S., Archibald N., Comber S., Knights A.M., Thompson R.C. & Firth L.B. (2018) Partial replacement of cement for waste aggregates in concrete coastal and marine infrastructure: a foundation for ecological enhancement? *Ecological Engineering*, 120, 655–667.
Natanzi A.S., Thompson B.J., Brooks P.R., Crowe T.P. & McNally C. (2021) Influence of concrete properties on the initial biological colonisation of marine artificial structures. *Ecological Engineering*, 159, 106104.

A replicated, randomized, controlled study in 2008–2009 on two intertidal rocky reefs on open coastlines in the Celtic Sea and the English Channel, UK (1) found that limestone settlement plates supported lower microalgal abundance than concrete plates, while abundance on granite plates was higher than or similar to concrete depending on the type of microalgae. After eight months, round microalgal abundance was lower on limestone plates (5% cover) than concrete (61%), and higher than both on granite plates (82%). Filamentous microalgae was less abundant on limestone (13%) than granite (33%) and concrete (30%), which were similar. Settlement plates (100 × 100 mm) were made from limestone, granite and concrete. Two of each were randomly arranged horizontally in each of two patches at midshore on each of two rocky reefs in May 2008. Microalgal cover on plates was measured using a scanning electron microscope after eight months.

A replicated, randomized, controlled study in 2007–2008 in four intertidal boulder-fields in Sydney Harbour estuary, Australia (2) found that using sandstone boulders in place of basalt boulders altered the macroalgae and non-mobile invertebrate community composition in two of four sites, and that abundances varied depending on the species group and site. After 10 months, macroalgae and non-mobile invertebrate community composition differed on sandstone and basalt boulders in two of four sites, but was similar in the other two sites (data reported as statistical model results). Sandstone boulders supported higher non-turf macroalgal abundance (0–17% cover) than basalt boulders (0–10%), and higher turf macroalgal abundance at one site (sandstone: 48%; basalt: 1%), but similar turf abundance at the other three sites (14–31 vs 9–25%).

Sandstone boulders supported similar abundances of tubeworms (Serpulidae) and oysters (Ostreidae) to basalt boulders (tubeworms: 7–24 vs 8–27%; oysters: 0–9 vs 1–9%), but fewer barnacles (Cirripedia) (0 vs 1–2%). Five sandstone and five basalt oval quarried boulders (diameter: 350 mm) were randomly arranged at lowshore in each of two basalt (artificial) and two sandstone (unspecified) boulder-fields in June 2007. Macroalgae and non-mobile invertebrates were counted on boulders over 10 months.

A replicated, randomized, controlled study in 2014–2015 on an intertidal rocky reef on open coastline in the Irish Sea, UK (3) found that hemp-concrete and shell-concrete settlement plates supported higher macroalgae and invertebrate cover than standard-concrete plates, and that hemp-concrete supported higher species richness than shell- and standard-concrete plates, with different community composition to standard-concrete plates. After 12 months, macroalgae and non-mobile invertebrate cover was similar on hemp-concrete (92% cover) and shell-concrete (74%) plates, and higher on both than standard-concrete plates (25%). Mobile invertebrate species richness was higher on hemp-concrete (8 species groups/plate) than shell-concrete (4/plate) and standard-concrete (3/plate), which were similar. Macroalgae and non-mobile invertebrate species richness was similar on all materials (hemp: 7/plate; shell: 6/plate; standard: 5/plate). Macroalgae and invertebrate community composition differed on hemp-concrete and standard-concrete, but shell-concrete was similar to both (data reported as statistical model results). Settlement plates (150 × 150 mm) were moulded from hemp-concrete, shell-concrete and standard-concrete. Five of each were randomly arranged horizontally at mid-lowshore on a rocky reef in October 2014. Macroalgae and invertebrates on plates were counted in the laboratory after 12 months.

A replicated, randomized, paired sites, controlled, before-and-after study in 2014–2016 on an intertidal seawall in a marina in the Mediterranean Sea, Israel (4) found that seawall panels made from ECOncrete™, along with grooves, small ledges and holes created on them, supported higher macroalgae and invertebrate species diversity and richness and different community composition compared with standard-concrete seawall surfaces without added habitats. After 22 months, macroalgae and invertebrate species diversity (data reported as Shannon index) and richness was higher on ECOncrete™ panels with added habitats (8 species/quadrat) than on standard-concrete seawall surfaces without (3/quadrat), and compared with seawall surfaces before panels were attached (2/quadrat). Community composition differed between ECOncrete™ panels and standard-concrete surfaces (data reported as statistical model results). Five species groups (1 macroalgae, 4 non-mobile invertebrates) recorded on panels were absent from standard-concrete surfaces. It is not clear whether these effects were the direct result of using environmentally-sensitive material or creating grooves, ledges and/or holes. Seawall panels (height: 1.5 m; width: 0.9 m; thickness: 130 mm) were made from patented ECOncrete™ material using a formliner. Panels had multiple grooves, small ledges and holes. Four panels were attached to a vertical concrete seawall in November 2014. The top 0.3 m were intertidal. Panels were compared with standard-concrete seawall surfaces cleared of organisms (height: 0.3 m; width: 0.9 m) adjacent to each panel.

Macroalgae and invertebrates were counted in one 300 × 300 mm randomly-placed quadrat on each panel and seawall surface during high tide over 22 months.

A replicated, randomized, paired sites, controlled study (year not reported) on an intertidal seawall in Ceuta Port in the Alboran Sea, Spain (5) found that sandstone settlement plates had higher chlorophyll-a and diatom abundance than limestone, slate, gabbro and concrete plates, and that material altered the diatom community composition but not their species richness or diversity. After two months, chlorophyll-a density was higher on sandstone settlement plates (18 $\mu g/cm^2$) than limestone (3 $\mu g/cm^2$), slate (3 $\mu g/cm^2$) and concrete (6 $\mu g/cm^2$) plates, which were all similar, while gabbro plates were similar to all materials (13 $\mu g/cm^2$). Diatom species diversity and richness (data not reported) was similar on all materials, while their community composition differed (data reported as statistical model results), but it was not clear which materials differed. Total diatom abundance was higher on sandstone plates (841 individuals) than limestone (329), slate (104), gabbro (275) and concrete (173). Settlement plates (170 × 170 mm) were made from sandstone, limestone, slate, gabbro and concrete. One of each was randomly arranged horizontally on each of five midshore boulders along a limestone boulder seawall (month/year not reported). Plate surfaces had grooves and small protrusions created on them. Microalgae and chlorophyll-a on plates were measured using a scanning electron microscope and spectrophotometer, respectively, after two months.

A replicated, randomized, controlled study in 2016–2017 on three intertidal seawalls in the Clyde and Forth estuaries and on open coastline in the English Channel, UK (6) found that using limestone-cement in place of concrete in settlement plates, along with creating pits, grooves, small ridges and textured surfaces, had mixed effects on macroalgae and invertebrate species richness and invertebrate abundances on plates, depending on the site. After 18 months, in three of six comparisons, macroalgae and mobile invertebrate species richness was higher on limestone-cement settlement plates with added habitats (2 species/plate) than concrete plates without (1/plate). In four of six comparisons, the same was true for mobile invertebrate abundance (limestone-cement: 4–11; concrete: 1–2 individuals/plate) and barnacle (Cirripedia) cover (48–74 vs 22–34%). In the other comparisons, no significant effects were found for richness (3 comparisons: 1–2 vs 1/plate), mobile abundances (2 comparisons: 1–2 vs 2–3/plate) or barnacle cover (2 comparisons: 46–84 vs 22–83%). It is not clear whether these effects were the direct result of using environmentally-sensitive material or creating pits, grooves, ridges and/or texture. Settlement plates (150 × 150 mm) were moulded from limestone-cement or concrete. Limestone-cement plates had pits, grooves and ridges, or textured surfaces, while concrete plates did not. Eight plates of each limestone-cement design were randomly arranged at upper-midshore on each of two vertical concrete seawalls in April–May 2016. Eight concrete plates were attached on both walls plus one other. Macroalgae and invertebrates on plates were counted from photographs over 18 months.

A replicated, controlled study in 2018–2019 on four intertidal seawalls on island coastlines in the Singapore Strait, Singapore, and in the Plym and Tamar estuaries, UK (7) found that reducing the pH of concrete settlement plates did not alter the macroalgae and invertebrate community composition or increase their species richness or abundance on plates. Over 12 months, reduced-pH-concrete settlement plates supported 59 invertebrate species in total (Singapore: 46; UK: 13), while standard-concrete plates supported 57 (Singapore: 48; UK: 9) (data not statistically tested). Ten invertebrate species (8 mobile, 2 non-mobile) recorded on reduced-pH plates were absent from standard-concrete plates. After 12 months, macroalgae and invertebrate community composition (data reported as statistical model results) and species richness was similar on reduced-pH plates (3–21 species/plate) and standard-concrete plates (3–20/plate). The same was true for invertebrate abundance (6–187 vs 11–216 individuals/plate) and cover of limpets (Patellidae, Fissurellidae, Siphonariidae, Lottioidea) (both 1–5% cover), barnacles (Cirripedia) (18–24 vs 18–25%), ephemeral green macroalgae (4–5 vs 5–8%) and encrusting macroalgae (35 vs 29%). Concrete settlement plates (200 × 200 mm) were moulded with reduced pH (pH 7–10) and standard pH (pH 12–13). Twenty-four of each were attached at a 60° angle at midshore on each of two seawalls in both Singapore and the UK during February–March 2018. Plates had water-retaining pits created on them. Macroalgae on plates were counted from photographs and invertebrates in the laboratory over 12 months. Eight plates were missing and no longer provided habitat.

A replicated, randomized, controlled study in 2018 on an intertidal breakwater on open coastline in the Irish Sea, Ireland (8) found that replacing standard Portland-cement with Ground Granulated Blast-Furnace Slag (GGBS), limestone-aggregate with granite-aggregate, and omitting plasticiser in concrete settlement plates had mixed effects on microalgal and barnacle (Cirripedia) abundances, depending on the material combination, wave-exposure and species group. After one month, on the wave-sheltered side of the breakwater, microalgal biomass was higher on plates with GGBS-cement (0.14–2.48 μg/cm^2) than standard-cement (0.03–0.74 μg/cm^2). Barnacle abundance varied depending on the aggregate and presence of plasticiser (GGBS-cement: 316–2,961 individuals/plate; standard-cement: 603–1,869/plate). There was no significant difference in microalgal or barnacle abundance between plates with granite-aggregate (microalgae: 0.03–1.66 μg/cm^2; barnacles: 316–2,961/plate) and limestone-aggregate (microalgae: 0.06–2.48 μg/cm^2; barnacles: 973–2,263/plate), or between plates without and with plasticiser (microalgae: 0.06–2.48 vs 0.03–1.66 μg/cm^2; barnacles: 316–2,263 vs 603–2,961/plate). On the exposed side of the breakwater, results varied depending on the cement-aggregate-plasticiser combination and species group. Concrete settlement plates (200 × 200 mm) were moulded with different cement (GGBS, standard Portland-cement), aggregates (granite, limestone) and additives (no plasticiser, plasticiser). Six plates of each binder-aggregate-additive combination were randomly arranged vertically at mid-lowshore on the wave-sheltered side of a boulder breakwater in April 2018. Two plates of each were attached on the wave-exposed side. Microalgal biomass on plates was measured using a fluorometer and barnacles were counted from photographs after 1 month.

(1) Coombes M.A., Naylor L.A., Thompson R.C., Roast S.D., Gómez-Pujol L. & Fairhurst R.J. (2011) Colonization and weathering of engineering materials by marine microorganisms: an SEM study. *Earth Surface Processes and Landforms*, 36, 582–593.
(2) Green D.S., Chapman M.G. & Blockley D.J. (2012) Ecological consequences of the type of rock used in the construction of artificial boulder-fields. *Ecological Engineering*, 46, 1–10.
(3) Dennis H.D., Evans A.J., Banner A.J. & Moore P.J. (2018) Reefcrete: reducing the environmental footprint of concretes for eco-engineering marine structures. *Ecological Engineering*, 120, 668–678.
(4) Perkol-Finkel S., Hadary T., Rella A., Shirazi R. & Sella I. (2018) Seascape architecture – incorporating ecological considerations in design of coastal and marine infrastructure. *Ecological Engineering*, 120, 645–654.
(5) Sempere-Valverde J., Ostalé-Valriberas E., Farfán G.M. & Espinosa F. (2018) Substratum type affects recruitment and development of marine assemblages over artificial substrata: a case study in the Alboran Sea. *Estuarine, Coastal and Shelf Science*, 204, 56–65.
(6) MacArthur M., Naylor L.A., Hansom J.D., Burrows M.T., Loke L.H.L. & Boyd I. (2019) Maximising the ecological value of hard coastal structures using textured formliners. *Ecological Engineering: X*, 1, 100002.
(7) Hsiung A.R., Tan W.T., Loke L.H.L., Firth L.B., Heery E.C., Ducker J., Clark V., Pek Y.S., Birch W.R., Ang A.C.F., Hartanto R.S., Chai T.M.F. & Todd P.A. (2020) Little evidence that lowering the pH of concrete supports greater biodiversity on tropical and temperate seawalls. *Marine Ecology Progress Series*, 656, 193–205.
(8) Natanzi A.S., Thompson B.J., Brooks P.R., Crowe T.P. & McNally C. (2021) Influence of concrete properties on the initial biological colonisation of marine artificial structures. *Ecological Engineering*, 159, 106104.

2.2. Create textured surfaces (≤1 mm) on intertidal artificial structures

- **Four studies** examined the effects of creating textured surfaces on intertidal artificial structures on the biodiversity of those structures. Two studies were on open coastlines in the UK[1] and the Netherlands[2a], one was in a port in the Netherlands[2b], and one was on an open coastline and in estuaries in the UK[3].

COMMUNITY RESPONSE (1 STUDY)

- **Overall richness/diversity (1 study):** One replicated, randomized, controlled study in the UK[3] found that creating textured surfaces on intertidal artificial structures, along with using environmentally-sensitive material, had mixed effects on the combined macroalgae and invertebrate species richness on structure surfaces, depending on the type of texture created and the site.

POPULATION RESPONSE (4 STUDIES)

- **Algal abundance (2 studies):** Two replicated, paired sites, controlled studies in the Netherlands[2a,2b] reported that creating textured surfaces on intertidal artificial structures did not increase the macroalgal abundance on structure surfaces.

- **Invertebrate abundance (4 studies):** Two of four replicated, controlled studies (including two randomized and two paired sites studies) in the UK[1,3] and the Netherlands[2a,2b] reported that creating textured surfaces on intertidal artificial structures did not increase the invertebrate abundance on structure surfaces[2a,2b]. One study[3] found that creating textured surfaces, along with using environmentally-sensitive material, had mixed effects on barnacle and mobile invertebrate abundances, depending on the site. One[1] found increased barnacle abundance, regardless of the type of texture created, but that different textures supported different abundances.

BEHAVIOUR (0 STUDIES)

Background

Texture influences the settlement and survival of marine organisms in intertidal rocky habitats. It provides secure anchor points for invertebrate larvae and algal germlings, helping them to resist dislodgement and escape predation or grazing (Lubchenco 1983). Settlement preferences and competitive interactions lead to some species being more abundant than others on textured surfaces (Harlin & Lindbergh 1977). These patterns vary by species, environmental conditions and the match or mismatch between the size and shape of the texture and organisms (Wahl & Hoppe 2002).

Most substrates have some form of texture, but marine artificial structures often have smoother surface texture than natural rocky substrates (Sedano *et al.* 2020). Structures with rougher texture tend to be more-readily colonized by invertebrates and algae (Moschella *et al.* 2005; Sempere-Valverde *et al.* 2018; but see Cacabelos *et al.* 2016), promoting community development. Textured surfaces can be created on intertidal artificial structures by moulding or treating surfaces during construction or retrospectively. Texture can also be altered indirectly through material choice. Studies that examine the effects of using alternative materials with incidentally-different textures are not considered here, but are included under the action "*Use environmentally-sensitive material on intertidal artificial structures*".

There are bodies of literature investigating the effects of textured surfaces on recruitment and community development in intertidal rocky habitats (e.g. Dudgeon & Petraitis 2005; van Tamelin *et al.* 1997), laboratory-based settlement behaviour (e.g. Neo *et al.* 2009), and also the use of micro-texture for anti-fouling applications (reviewed by Scardino & de Nys 2011). These studies are not included in this synopsis, which focusses on *in situ* conservation actions to promote colonization of biodiversity on marine artificial structures.

Definition: 'Texture' is micro-scale roughness applied to an entire surface that produces depressions and/or elevations ≤1 mm (Strain *et al.* 2018).

See also: *Use environmentally-sensitive material on intertidal artificial structures; Create natural rocky reef topography on intertidal artificial structures*; *Create pit habitats (1–50 mm) on intertidal artificial structures*; *Create groove habitats (1–50 mm) on intertidal artificial structures*; *Create small protrusions (1–50 mm) on intertidal artificial structures; Create small ridges or ledges (1–50 mm) on intertidal artificial structures*; *Create groove habitats* and *small protrusions, ridges or ledges (1–50 mm) on intertidal artificial structures*.

Cacabelos E., Martins G.M., Thompson R., Prestes A.C.L., Azevedo J.M.N. & Neto A.I. (2016) Material type and roughness influence structure of inter-tidal communities on coastal defences. *Marine Ecology*, 37, 801–812.
Dudgeon S. & Petraitis P.S. (2005) First year demography of the foundation species, *Ascophyllum nodosum*, and its community implications. *Oikos*, 109, 405–415.
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Lubchenco J. (1983) *Littorina* and *Fucus*: effects of herbivores, substratum heterogeneity, and plant escapes during succession. *Ecology*, 64, 1116–1123.
Moschella P.S., Abbiati M., Åberg P., Airoldi L., Anderson J.M., Bacchiocchi F., Bulleri F., Dinesen G.E., Frost M., Gacia E., Granhag L., Jonsson P.R., Satta M.P., Sundelöf A., Thompson R.C. & Hawkins S.J. (2005) Low-crested coastal defence structures as artificial habitats for marine life: using ecological criteria in design. *Coastal Engineering*, 52, 1053–1071.
Neo M.L., Todd P.A., Teo S.L.-M. & Chou L.M. (2009) Can artificial substrates enriched with crustose coralline algae enhance larval settlement and recruitment in the fluted giant clam (*Tridacna squamosa*)? *Hydrobiologia*, 625, 83–90.
Scardino A.J. & de Nys R. (2011) Mini review: biomimetic models and bioinspired surfaces for fouling control. *Biofouling: The Journal of Bioadhesion and Biofilm Research*, 27, 73–86.
Sedano F., Navarro-Barranco C., Guerra-García J.M. & Espinosa F. (2020) Understanding the effects of coastal defence structures on marine biota: the role of substrate composition and roughness in structuring sessile, macro- and meiofaunal communities. *Marine Pollution Bulletin*, 157, 111334.
Sempere-Valverde J., Ostalé-Valriberas E., Farfán G.M. & Espinosa F. (2018) Substratum type affects recruitment and development of marine assemblages over artificial substrata: a case study in the Alboran Sea. *Estuarine, Coastal and Shelf Science*, 204, 56–65.
Strain E.M.A., Olabarria C., Mayer-Pinto M., Cumbo V., Morris R.L., Bugnot A.B., Dafforn K.A., Heery E., Firth L.B., Brooks P.R. & Bishop M.J. (2018) Eco-engineering urban infrastructure for marine and coastal biodiversity: which interventions have the greatest ecological benefit? *Journal of Applied Ecology*, 55, 426–441.
van Tamelen P.G., Stekoll M.S. & Deysher L. (1997) Recovery processes of the brown alga *Fucus gardneri* following the 'Exxon Valdez' oil spill: settlement and recruitment. *Marine Ecology Progress Series*, 160, 265–277.
Wahl M. & Hoppe K. (2002) Interactions between substratum rugosity, colonization density and periwinkle grazing efficiency. *Marine Ecology Progress Series*, 225, 239–249.

A replicated, randomized, controlled study in 2010 on two intertidal rocky reefs on open coastlines in the Celtic Sea and the English Channel, UK (1) found that creating textured surfaces on settlement plates increased the abundance of barnacles *Chthamalus* spp. on plates. After six months, average barnacle abundance was higher on scrape-textured plates (226–351/plate) than spray-textured plates (124–228/plate), and higher on both than on untextured plates (59–152/plate). Concrete settlement plates (50 × 50 mm) were made with and without textured surfaces, created by scraping with a wire brush or spraying with a water jet. Ten plates with each of 'scrape-textured', 'spray-textured' and untextured surfaces were randomly arranged horizontally at midshore on each of two rocky reefs in May 2010. Barnacles on plates were counted from photographs after six months.

A replicated, paired sites, controlled study in 2008–2010 on an intertidal breakwater on open coastline in the North Sea, Netherlands (2a) reported that settlement plates with textured surfaces supported similar abundances of macroalgae and invertebrates to plates without texture. Data were not statistically tested. After 28 months, there were no clear differences in macroalgal or invertebrate abundances on plates with and without textured surfaces (data not reported). Concrete settlement plates (250 × 250 mm) were made with and without textured surfaces using a mould. Plates with texture had either fine (0.5 mm) or coarse (1 mm) texture. One of each and one plate without texture were placed on each of 10 vertical surfaces on each side of a concrete-block breakwater (wave-exposed, wave-sheltered) in May 2008. One plate with fine texture and one without were also placed on each of 10 horizontal surfaces on each side of the breakwater. On the wave-exposed side, plates were at mid-highshore, while on the wave-sheltered side, plates were at low-midshore. Macroalgae and invertebrates on plates were counted during low tide over 28 months.

A replicated, paired sites, controlled study in 2009 on 14 jetty pilings in Rotterdam Port in the Rhine-Meuse estuary, Netherlands (2b) reported that settlement plates with textured surfaces supported similar abundances of macroalgae and invertebrates to plates without texture. Data were not statistically tested. After nine months, there were no clear differences in macroalgal or invertebrate abundances on plates with and without textured surfaces (data not reported). Concrete settlement plates (250 × 250 mm) were made with and without textured surfaces using a mould. One plate with texture and one without were attached to vertical surfaces on each of 14 wooden pilings at lowshore in March 2009. Macroalgae and invertebrates on plates were counted during low tide over nine months.

A replicated, randomized, controlled study in 2016–2017 on three intertidal seawalls in the Clyde and Forth estuaries and on open coastline in the English Channel, UK (3) found that creating textured surfaces on seawall surfaces, along with using environmentally-sensitive material, had mixed effects on macroalgae and invertebrate species richness and invertebrate abundances, depending on the type of texture created and the site. After 18 months, plates with and without texture supported similar macroalgae and mobile invertebrate species richness in seven of eight comparisons (textured: 1–2 species/plate; untextured: 1/plate). At one site (1 comparison), cast-textured plates supported more species (2/plate) than untextured plates (1/plate). Textured and untextured plates also supported similar mobile invertebrate abundance in five of eight comparisons (textured: 1–2 individuals/plate; untextured: 1–3/plate). At one site (3 comparisons), textured plates supported more mobile invertebrates (3–5 individuals/plate) than untextured plates (1/plate). Barnacle (Cirripedia) cover was higher on plates with texture (67–95%) than without (22–83%) in six of eight comparisons, but did not significantly differ at one site (2 comparisons; textured: 46–51%; untextured: 22%). It is not clear whether these effects were the direct result of creating texture or using environmentally-sensitive material on some plates. Settlement plates (150 × 150 mm) were made with and without textured surfaces, created by scraping with a wire brush, moulding with barnacle-shaped impressions, or casting with crushed foil. 'Scrape-textured', 'mould-textured' and untextured plates were concrete, while 'cast-textured' plates were limestone-cement (environmentally-sensitive material). Eight plates with each of scrape-textured, mould-textured and untextured surfaces were randomly arranged at upper-midshore on each of three vertical concrete seawalls in April–May 2016. Eight cast-textured plates were attached on each of two walls. Macroalgae and invertebrates on plates were counted from photographs over 18 months.

(1) Coombes M.A., La Marca E.C., Naylor L.A. & Thompson R.C. (2015) Getting into the groove: opportunities to enhance the ecological value of hard coastal infrastructure using fine-scale surface textures. *Ecological Engineering*, 77, 314–323.
(2) Paalvast P. (2015) *The role of geometric structure and texture on concrete for algal and macrofaunal colonization in the marine and estuarine intertidal zone.* Proceedings of RECIF Conference on artificial reefs: From materials to ecosystems. Caen, France, 77–84.
(3) MacArthur M., Naylor L.A., Hansom J.D., Burrows M.T., Loke L.H.L. & Boyd I. (2019) Maximising the ecological value of hard coastal structures using textured formliners. *Ecological Engineering: X*, 1, 100002.

2.3. Create natural rocky reef topography on intertidal artificial structures

- **Two studies** examined the effects of creating natural rocky reef topography on intertidal artificial structures on the biodiversity of those structures. One study was on an open coastline and in estuaries in the UK[1], and one was on an open coastline in the UK[2].

COMMUNITY RESPONSE (1 STUDY)

- **Overall richness/diversity (1 study):** One replicated, randomized, controlled study in the UK[1] found that creating natural rocky reef topography on intertidal artificial structures did not increase the combined macroalgae and invertebrate species richness on structure surfaces.

POPULATION RESPONSE (1 STUDY)

- **Invertebrate abundance (1 study):** One replicated, randomized, controlled study in the UK[1] found that creating natural rocky reef topography on intertidal artificial structures had mixed effects on barnacle and mobile invertebrate abundances on structure surfaces, depending on the site.

BEHAVIOUR (1 STUDY)

- **Use (1 study):** One study in the UK[2] reported that natural topography created on intertidal artificial structures was colonized by macroalgae and limpets, and that limpets used shaded grooves and water-retaining depressions created by the topography.

Background

Topography influences the settlement and survival of marine organisms on intertidal rocky substrates. Variation in topography generates variation in the physical environment and plays an important role in sustaining biodiversity and ecological functioning (Levin 1974). On rocky reefs, many habitat features that offer refuge from physical stressors and predation, such as bumps, crevices and holes, are generated as a function of substrate topography and geomorphology. The full fingerprint of natural rocky reef topography encompasses a variety of habitat features of different scales interacting within a mosaic.

Marine artificial structures often have much lower topographic variability than natural rocky reefs, which is thought to be a key reason for their reduced biodiversity (Firth *et al.* 2013; Moschella *et al.* 2005). Natural rocky reef topography can be created on intertidal artificial structures by moulding or casting material during construction or retrospectively (see Evans *et al.* 2021).

Definition: 'Natural rocky reef topography' refers to the full fingerprint of substrate topography found in natural rocky habitats.

See also: *Create textured surfaces (≤1 mm) on intertidal artificial structures*; *Create pit habitats (1–50 mm) on intertidal artificial structures*; *Create hole habitats (>50 mm) on intertidal artificial structures*; *Create groove habitats (1–50 mm) on intertidal artificial structures*; *Create crevice habitats (>50 mm) on intertidal artificial structures*; *Create 'rock pools' on intertidal artificial structures*; *Create small protrusions (1–50 mm) on intertidal artificial structures; Create large protrusions (>50 mm) on intertidal artificial structures; Create small ridges or ledges (1–50 mm) on intertidal artificial structures*; *Create large ridges or ledges (>50 mm) on intertidal artificial structures*; *Create groove habitats* and *small protrusions, ridges or ledges (1–50 mm) on intertidal artificial structures*.

Evans A.J., Lawrence P.J., Natanzi A.S., Moore P.J., Davies A.J., Crowe T.P., McNally C., Thompson B., Dozier A.E. & Brooks P.R. (2021) Replicating natural topography on marine artificial structures – a novel approach to eco-engineering. *Ecological Engineering*, 160, 106144.

Firth L.B., Thompson R.C., White F.J., Schofield M., Skov M.W., Hoggart S.P.G., Jackson J., Knights A.M. & Hawkins S.J. (2013) The importance of water-retaining features for biodiversity on artificial intertidal coastal defence structures. *Diversity and Distributions*, 19, 1275–1283.

Levin S.A. (1974) Dispersion and population interactions. *American Society of Naturalists*, 108, 207–228.

Moschella P.S., Abbiati M., Åberg P., Airoldi L., Anderson J.M., Bacchiocchi F., Bulleri F., Dinesen G.E., Frost M., Gacia E., Granhag L., Jonsson P.R., Satta M.P., Sundelöf A., Thompson R.C. & Hawkins S.J. (2005) Low-crested coastal defence structures as artificial habitats for marine life: using ecological criteria in design. *Coastal Engineering*, 52, 1053–1071.

A replicated, randomized, controlled study in 2016–2017 on three intertidal seawalls in the Clyde and Forth estuaries and on open coastline in the English Channel, UK (1) found that creating natural rocky reef topography on the seawalls did not increase the macroalgae and invertebrate species richness on seawall surfaces, but increased invertebrate abundances at one of three sites. After 18 months, macroalgae and mobile invertebrate species richness was similar on settlement plates with and without natural rocky reef topography (both 1 species/plate). Barnacle (Cirripedia) and mobile invertebrate abundances were higher on plates with topography than without at one site (barnacles: 72 vs 34% cover; mobiles: 3 vs 1 individuals/plate), but were statistically similar at two sites (barnacles: 48–93 vs 22–83%; mobiles: 1 vs 2–3/plate). Concrete settlement plates (150 × 150 mm) were made with and without natural rocky reef topography moulded from digital scans of a natural boulder surface. Eight plates with topography and eight without were randomly arranged at upper-midshore on each of three vertical concrete seawalls in April–May 2016. Macroalgae and invertebrates on plates were counted from photographs over 18 months.

A study in 2019 on two intertidal breakwaters on open coastline in the Irish Sea, UK (2) reported that natural rocky reef topography created on the breakwaters supported macroalgae (*Ulva* spp.) and limpets (*Patella* spp.). Over four months, green macroalgae and adult and juvenile limpets were recorded on settlement plates with natural rocky reef topography. Limpets were seen using shaded grooves and water-retaining depressions created by the natural topography. Concrete settlement plates (250 × 250 mm) were made with natural rocky reef topography moulded from digital scans of natural reef surfaces. Natural surfaces were selected based on the biodiversity they supported and measured features of the underlying topography. They were designed to target high species richness, rare species, or species that were common on natural reefs but not on artificial structures. Plates with natural topography were attached on horizontal surfaces of two granite boulder breakwaters in August 2019 (A. Evans, *pers. comms.*). Macroalgae and invertebrates on plates were counted during low tide over four months.

(1) MacArthur M., Naylor L.A., Hansom J.D., Burrows M.T., Loke L.H.L. & Boyd I. (2019) Maximising the ecological value of hard coastal structures using textured formliners. *Ecological Engineering: X*, 1, 100002.
(2) Evans A.J., Lawrence P.J., Natanzi A.S., Moore P.J., Davies A.J., Crowe T.P., McNally C., Thompson B., Dozier A.E. & Brooks P.R. (2021) Replicating natural topography on marine artificial structures - a novel approach to eco-engineering. *Ecological Engineering*, 160, 106144.

2.4. Create pit habitats (1–50 mm) on intertidal artificial structures

- **Twenty-two studies** examined the effects of creating pit habitats on intertidal artificial structures on the biodiversity of those structures. Ten studies were on open coastlines in the UK[1,4,5a,5c,11a,11b], the Netherlands[6a] and the Azores[2,9,12], six were on island coastlines in the Singapore Strait[7,8a,8b,10,13,16], three were in estuaries in southeast Australia[3] and the UK[5b,14b], one was in a port in the Netherlands[6b], one was in an estuary and on an open coastline in the UK[14a], and one was on island coastlines in the Singapore Strait and in estuaries in the UK[15].

COMMUNITY RESPONSE (16 STUDIES)

- **Overall community composition (9 studies):** Four of six replicated, controlled studies (including four randomized and two before-and-after studies) in Australia[3], Singapore[8a,8b,10] and the UK[11a,11b] found that creating pit habitats on intertidal artificial structures altered the combined macroalgae and invertebrate community composition on structure surfaces[8a,10,11a,11b]. One study[3] found that creating pits did not alter the community composition. One[8b] found that creating pits, along with grooves, small protrusions and ridges, had mixed effects depending on the size and arrangement of pits and other habitats and the site, while one[10] found that varying the pit size and arrangement had no significant effect. Three of these studies[10,11a,11b], along with three other replicated, controlled studies (including one that was randomized) in the UK[1,5a] and Singapore[7], reported that pit habitats, along with grooves and ridges in one[7], supported macroalgae[7,11a], invertebrate[5a,11b] and/or fish[11b] species that were absent from structure surfaces without added habitats.
- **Fish community composition (1 study):** One replicated, randomized, controlled study in Singapore[16] found that pit habitats created on an intertidal artificial structure, along with grooves,

altered the fish community composition on and around structure surfaces, and supported species that were absent from surfaces without pits and grooves.

- **Overall richness/diversity (12 studies):** Eight of 12 replicated controlled studies (including six randomized and two before-and-after studies) in the UK[1,5a,5b,5c,11a,11b,14a,14b] and Singapore[8a,8b,10,13] found that creating pit habitats on intertidal artificial structures, along with grooves[13], or grooves, small protrusions and ridges[8b] in two studies, increased the combined macroalgae and invertebrate species richness[1,5a,8a,8b,10,11a,11b,13] and/or diversity[11a,11b] on structure surfaces. Two studies[5b,5c] found that creating pits did not increase the species richness, while two[14a,14b] found that creating pits, along with grooves[14b] or using environmentally-sensitive material[14a], had mixed effects depending on the site. One of the studies[10] found that varying the pit size and arrangement resulted in higher species richness, while one[8b] found that this had mixed effects depending on the shore level. Two of the studies[5a,5c] found that varying the pit size did not affect species richness. One of them[13] found that increasing the density and fragmentation of pits, along with grooves, had mixed effects on species richness.
- **Algal richness/diversity (1 study):** One replicated, randomized, controlled study in Singapore[7] reported that creating pits on an intertidal artificial structure, along with grooves and small ridges, increased the macroalgal species richness on structure surfaces.
- **Invertebrate richness/diversity (2 studies):** One of two replicated, randomized, controlled studies in Australia[3] and the Azores[12] reported that creating pits on an intertidal artificial structure increased the limpet and periwinkle species richness on structure surfaces, and that their richness and diversity varied depending on the pit arrangement[12]. One[3] found that creating pits did not affect the limpet species richness, regardless of the pit size.
- **Fish richness/diversity (1 study):** One replicated, randomized, controlled study in Singapore[16] found that creating pit habitats on an intertidal artificial structure, along with grooves, increased the fish species richness on and around structure surfaces.

POPULATION RESPONSE (15 STUDIES)

- **Overall abundance (5 studies):** Two of five replicated, controlled studies (including three randomized and two before-and-after studies) in Singapore[8a,8b,16] and the UK[11a,11b] found that creating pit habitats on intertidal artificial structures, along with grooves in one study[16], increased the combined macroalgae and invertebrate abundance on structure surfaces[11a,16]. One study[11b] found that creating pits decreased their abundance and one[8a] found no effect. One[8b] found that creating pits, along with grooves, small protrusions and ridges, had mixed effects on abundance depending on the pit size and arrangement, shore level and site.
- **Algal abundance (4 studies):** Three of four replicated, controlled studies (including two randomized and two paired sites studies) in the Netherlands[6a,6b], Singapore[7] and the Azores[9] found that creating pit habitats on intertidal artificial structures, along with grooves and small ridges in one study[7], did not increase the macroalgal abundance on structure surfaces[6a,6b,7]. One study[9] found that creating pits had mixed effects on abundance depending on the pit size and arrangement and the site.
- **Invertebrate abundance (9 studies):** Three of eight replicated, controlled studies (including six randomized and two paired sites studies) in the Azores[2,9,12], the Netherlands[6a,6b], Australia[3] and the UK[14a,14b] found that creating pit habitats on intertidal artificial structures did not increase the

combined invertebrate[6a,6b] or mobile invertebrate[3] abundance on structure surfaces. Three studies[2,9,14b] found that creating pits, along with grooves in one study[14b], had mixed effects on barnacle[9,14b] and/or mobile invertebrate[2,9,14b] abundances, depending on the site[2,9,14b], the species[9], the size of animals[2], and/or the pit size and arrangement[9]. Two studies[12,14a] found that creating pits, along with using environmentally-sensitive material in one[14a], increased barnacle[14a] and/or mobile invertebrate[12,14a] abundances. Two of the studies found that the pit size[3] or arrangement[12] did not affect abundances, while two found that the effects of pit size and arrangement varied depending on the site[2,9] and species[9]. One replicated randomized study in the UK[4] found that increasing pit density increased periwinkle abundance, but pit arrangement did not.

- **Fish abundance (1 study):** One replicated, randomized, controlled study in Singapore[16] found that creating pit habitats on an intertidal artificial structure, along with grooves, increased the fish abundance on and around structure surfaces.

BEHAVIOUR (6 STUDIES)

- **Use (5 studies):** Two replicated, randomized, controlled studies in the Azores[2,12] reported that occupancy of pit habitats created on intertidal artificial structures by limpets[2,12] and/or periwinkles[12] varied depending on the pit size[2] and arrangement[2,12], the size of animals[2], the species[12] and/or site[2]. Three replicated studies (including two paired sites, controlled studies) in the Netherlands[6a,6b] and in Singapore and the UK[15] reported that pit habitats were used by periwinkles[6a,6b], macroalgae and invertebrates[15].
- **Fish behaviour change (1 study):** One replicated, randomized, controlled study in Singapore[16] found that creating pit habitats on an intertidal artificial structure, along with grooves, increased the number of bites fishes took from structure surfaces.

Background

Pit habitats provide organisms refuge from desiccation and temperature fluctuations during low tide in intertidal rocky habitats (Williams & Morrit 1995). They also provide shelter from predation or grazing (Menge & Lubchenco 1981) and some species preferentially settle into them (Skinner & Coutinho 2005). The size and density of pits is likely to affect the size, abundance and variety of organisms that can use them. Small pits can provide refuge for small-bodied organisms but may exclude larger organisms, limit their growth and get rapidly filled-up (Firth *et al.* 2020). Large pits can be used by larger-bodied organisms but may not provide sufficient refuge from predators for smaller organisms.

Pits are sometimes present on boulders used in marine artificial structures as a result of quarrying processes (Hall *et al.* 2018), and can form on other structures through erosion. However, these are often filled or repaired during maintenance works (Moreira *et al.* 2007) and are absent from many structures (Martins *et al.* 2010). Pit habitats can be created on intertidal artificial structures by adding or removing material, either during construction or retrospectively.

There is a body of literature investigating the effects of pit habitats on ecological interactions and processes in intertidal rocky habitats (e.g. Beck 2000; Chapman & Underwood 1994). These studies are not included in this synopsis, which focusses on *in situ* conservation actions to enhance the biodiversity of marine artificial structures.

Definition: 'Pit habitats' are depressions with a length to width ratio ≤3:1 and depth 1–50 mm (Strain *et al.* 2018). Intertidal pits may or may not retain water during low tide.

See also: *Create textured surfaces (≤1 mm) on intertidal artificial structures*; *Create natural rocky reef topography on intertidal artificial structures*; *Create hole habitats (>50 mm) on intertidal artificial structures*; *Create groove habitats (1–50 mm) on intertidal artificial structures*; *Create crevice habitats (>50 mm) on intertidal artificial structures*; *Create groove habitats* and *small protrusions, ridges or ledges (1–50 mm) on intertidal artificial structures*.

Beck M.W. (2000) Separating the elements of habitat structure: independent effects of habitat complexity and structural components on rocky intertidal gastropods. *Journal of Experimental Marine Biology and Ecology*, 249, 29–49.

Chapman M.G. & Underwood A.J. (1994) Dispersal of the intertidal snail, *Nodolittorina pyramidalis*, in response to the topographic complexity of the substratum. *Journal of Experimental Marine Biology and Ecology*, 179, 145–169.

Firth L.B., Airoldi L., Bulleri F., Challinor S., Chee S.-Y., Evans A.J., Hanley M.E., Knights A.M., O'Shaughnessy K., Thompson R.C. & Hawkins S.J. (2020) Greening of grey infrastructure should not be used as a Trojan horse to facilitate coastal development. *Journal of Applied Ecology*, 57, 1762–1768.

Hall A.E., Herbert R.J.H., Britton J.R. & Hull S.L. (2018) Ecological enhancement techniques to improve habitat heterogeneity on coastal defence structures. *Estuarine, Coastal and Shelf Science*, 210, 68–78.

Martins G.M., Thompson R.C., Neto A.I., Hawkins S.J. & Jenkins S.R. (2010) Enhancing stocks of the exploited limpet *Patella candei* d'Orbigny via modifications in coastal engineering. *Biological Conservation*, 143, 203–211.

Menge B.A. & Lubchenco J. (1981) Community organization in temperate and tropical rocky intertidal habitats: prey refuges in relation to consumer pressure gradients. *Ecological Monographs*, 51, 429–450.

Moreira J., Chapman M.G. & Underwood A.J. (2007) Maintenance of chitons on seawalls using crevices on sandstone blocks as habitat in Sydney Harbour, Australia. *Journal of Experimental Marine Biology and Ecology*, 347, 134–143.

Skinner L.F. & Coutinho R. (2005) Effect of microhabitat distribution and substrate roughness on barnacle *Teraclita stalactifera* (Lamarck, 1818) settlement. *Brazilian Archives of Biology and Technology*, 48, 109–113.

Strain E.M.A., Olabarria C., Mayer-Pinto M., Cumbo V., Morris R.L., Bugnot A.B., Dafforn K.A., Heery E., Firth L.B., Brooks P.R. & Bishop M.J. (2018) Eco-engineering urban infrastructure for marine and coastal biodiversity: which interventions have the greatest ecological benefit? *Journal of Applied Ecology*, 55, 426–441.

Williams G.A. & Morritt D. (1995) Habitat partitioning and thermal tolerance in a tropical limpet, *Cellana grata*. *Marine Ecology Progress Series*, 124, 89–103.

A replicated, controlled study in 2001–2002 on two intertidal breakwaters on open coastline in the English Channel, UK (1) reported that creating pit habitats on the breakwaters increased the macroalgae and invertebrate species richness on breakwater surfaces. After 12 months, settlement plates with pits supported five species in total, while plates without pits supported two species (data not statistically tested). Concrete settlement plates (300 × 300 mm) were made with and without pit habitats. Plates had six large (diameter: 30 mm) or 13 small (15 mm) round pits (depth: 20 mm), or a mixture of four large and four small pits (spacing/arrangement not reported). Four plates of each and four without pits were attached to horizontal midshore surfaces on each of two granite boulder breakwaters in 2001 (year: M. Hanley *pers. comms.*; month not reported). Macroalgae and invertebrates were counted on plates with and without pits during low tide after 12 months.

A replicated, randomized, controlled study in 2006–2007 on an intertidal seawall on open coastline in the Atlantic Ocean, Azores (2; same experimental set-up as 9) found that creating pit habitats on the seawall increased abundances of recently-recruited and juvenile limpets *Patella candei* at one of two sites, but not adults. After four months, at one of two sites, recruits and juveniles were more abundant on surfaces with pits (3–6 limpets/surface) than without (0–1/surface). At the second site, no recruits were recorded and juvenile abundance was similar on surfaces with and without pits (both 0/surface). At both sites, adult abundance was statistically similar on surfaces with pits (2–8/surface) and without (0–3/surface). At the first site, recruits occupying pits were more abundant in high-density pits (4–6/surface) than low (1–2/surface), while adults occupying pits were more abundant in large pits (9–11/surface) than small (1–2/surface). Pit habitats were created by drilling into a basalt boulder seawall in November 2006. Arrays of large (diameter: 24 mm) and small (12 mm) round pits (depth: 10 mm) were evenly-spaced on 250 × 250 mm seawall surfaces with high (16 pits/array) or low (8/array) densities. There were five surfaces with each size-density combination and five without pits, randomly arranged at midshore in each of two sites along the seawall. Limpets were removed from surfaces when pits were created, then were counted on surfaces with and without pits during low tide after four months.

A replicated, randomized, controlled study in 2000–2003 on an intertidal seawall in Sydney Harbour estuary, Australia (3) found that creating pit habitats on the seawall did not alter the macroalgae and invertebrate community composition or increase limpet (Patellidae and/or Siphonariidae, Fissurellidae) species richness or abundance, or chiton (Polyplacophora) abundance on seawall surfaces. After three months, seawall surfaces with pits supported similar macroalgae and invertebrate community composition to surfaces without (data reported as statistical model results). After 27 months, limpet species richness and abundance were similar in large pits (0 species and individuals/array), small pits (1 species/array, 2 individuals/array) and on surfaces without pits (1 species/surface, 3 individuals/surface). The same was true for chiton abundance (large pits: 0 individuals/array; small pits: 2/array; no pits: 0/surface). Pit habitats were created in 2000 (month not reported) by drilling into a vertical sandstone seawall during reconstruction. Large (diameter: 50 mm) and small (25 mm) round pits (depth: 5 mm) were drilled in arrays of 16 (spacing/arrangement not reported) on 1 × 0.4 m seawall surfaces. There were five surfaces with each of large, small and no pits, randomly arranged (shore level not reported). Macroalgae and invertebrates were counted on surfaces with and without pits during low tide after three months. Mobile invertebrates were counted in pits and on surfaces without after 27 months.

A replicated, randomized study in 2006–2008 on an intertidal seawall on open coastline in the English Channel, UK (4) found that pit habitats created on the seawall supported similar periwinkle *Melarhaphe neritoides* abundance regardless of the pit patchiness, but that increasing the pit density increased their abundance. Over 24 months, seawall surfaces with patchy pits supported similar periwinkle abundance (132–176 individuals/surface) to surfaces with evenly-spaced pits (170–208/surface). Abundance increased with increasing pit density (4 pits: 52 individuals/surface; 16 pits: 178/surface; 36 pits: 285/surface; 64 pits: 343/surface) but it was not clear which densities differed significantly from which. Pit habitats were created by drilling into a vertical concrete seawall in June 2006. Arrays of round pits (diameter: 10 mm; depth: 7 mm) were patchy (four patches/surface) or evenly-spaced on 500 × 500 mm seawall surfaces, with different densities (4, 16, 36 or 64 pits/surface). There were three surfaces with each arrangement-density combination randomly arranged at highshore. Existing cracks and holes were filled with cement and organisms were removed from surfaces when pits were created, then small periwinkles were counted on surfaces during low tide over 24 months.

A replicated, controlled study in 2011–2013 on an intertidal breakwater on open coastline in the English Channel, UK (5a) found that creating pit habitats on the breakwater increased the macroalgae and invertebrate species richness on breakwater surfaces. After 24 months, macroalgae and invertebrate species richness was similar on surfaces with large (10 species/surface) and small (9/surface) pits, and higher on both than on surfaces without pits (3/surface). Six invertebrate species groups recorded on surfaces with pits were absent from those without. Pit habitats were created in August 2011 by drilling into the vertical sides of concrete breakwater blocks. Arrays of 100 large (diameter: 22 mm) and small (14 mm) round pits (depth: 25 mm) were evenly-spaced on 1 × 1 m breakwater surfaces. There was one surface with each of large, small and no pits on each of eight blocks at mid-lowshore. Pits were angled to retain water. Macroalgae and invertebrates were counted on surfaces with and without pits during low tide after 24 months.

A replicated, controlled study in 2010–2011 on an intertidal seawall in the Teign estuary, UK (5b) found that pit habitats created on the seawall supported similar macroalgae and invertebrate species richness to seawall surfaces without pits. After 19 months, macroalgae and invertebrate species richness was similar in pits (2 species/array) and on surfaces without pits (1/surface). Pit habitats were created in May 2010 by pushing a stick into wet mortar between blocks during construction of a vertical sandstone seawall. Arrays of four round pits (diameter: 25 mm; depth: 25 mm) were evenly-spaced on 150 × 150 mm seawall surfaces. There were 15 surfaces with pits and 15 without at highshore. Pits were angled to retain water. Macroalgae and invertebrates were counted in pits and on surfaces without pits during low tide after 19 months. One array of pits and seven surfaces without had been buried by sediment and no longer provided habitat.

A replicated, controlled study in 2012–2013 on an intertidal groyne on open coastline in the Irish Sea, UK (5c) reported that pit habitats created on a concrete block placed in the groyne supported similar macroalgae and invertebrate species richness to groyne surfaces without pits. Data were not statistically tested. After 13 months, a total of three species were recorded in deep pits, two in shallow pits, and four on groyne surfaces without pits. Pit habitats were created on two vertical sides of a concrete block (1.5 × 1.5 × 1 m) using a mould. Two arrays of each of 16 deep (50 mm) and 16 shallow (20 mm) round pits (diameter: 20 mm) were evenly-spaced in 250 × 250 mm areas on each side. The block was placed at midshore in a boulder groyne during construction in February 2012. Surfaces with pits were compared with vertical surfaces of adjacent groyne boulders (dimensions/material not reported). Macroalgae and invertebrates were counted in pits and on groyne surfaces without during low tide after 13 months.

A replicated, paired sites, controlled study in 2008–2010 on an intertidal breakwater on open coastline in the North Sea, Netherlands (6a) reported that settlement plates with pit habitats supported similar abundances of macroalgae and invertebrates to plates without pits. Data were not statistically tested. After 28 months, there were no clear differences in macroalgal or invertebrate abundances on plates with and without pits (data not reported). Periwinkles *Littorina saxatilis* and *Littorina neritoides* were seen using pits. Concrete settlement plates (250 × 250 mm) were made with and without pit habitats using a mould. Plates with pits had 25 variable pits/plate (diameter: 12–35 mm; depth: 25–50 mm). One plate with pits and one without were placed on each of 10 horizontal and 10 vertical surfaces on each side of a concrete-block breakwater (wave-exposed, wave-sheltered) in May 2008. On the wave-exposed side, plates were at mid-highshore, while on the wave-sheltered side, plates were at low-midshore. Macroalgae and invertebrates on plates were counted during low tide over 28 months.

A replicated, paired sites, controlled study in 2009 on 14 jetty pilings in Rotterdam Port in the Rhine-Meuse estuary, Netherlands (6b) reported that settlement plates with pit habitats supported similar abundances of macroalgae and invertebrates to plates without pits. Data were not statistically tested. After nine months, there were no clear differences in macroalgal or invertebrate abundances on plates with and without pits (data not reported). Periwinkles *Littorina saxatilis* were seen using pits. Concrete settlement plates (250 × 250 mm) were made with and without pit habitats using a mould. Plates with pits had 25 variable pits/plate (diameter: 12–35 mm; depth: 25–50 mm). One plate with pits and one without were attached to vertical surfaces on each of 14 wooden pilings at lowshore in March 2009. Macroalgae and invertebrates on plates were counted during low tide over nine months.

A replicated, randomized, controlled study in 2011–2012 on two intertidal seawalls on island coastlines in the Singapore Strait, Singapore (7; same experimental set-up as 10) reported that concrete settlement plates with pit habitats, along with grooves and small ridges, supported higher macroalgal species richness but similar abundances compared with granite plates without added habitats. After 12 months, settlement plates with pits, grooves and ridges supported a total of five macroalgal species groups, while plates without supported three (data not statistically tested). Abundances of three species groups were statistically similar on plates with pits, grooves and ridges (18–41% cover) and without (5–61%) in five of six comparisons, while one group was more abundant on plates with pits, grooves and ridges (22–27 vs 5%) at one site. Abundances were similar on plates with variable (1–34%) and regular (3–41%) habitats. It is not clear whether these effects were the direct result of creating pits, grooves or ridges. Settlement plates (400 × 400 mm) were moulded with pit habitats, with grooves and small ridges, and with neither. Plates with pits, grooves and ridges were concrete with 36 square pits/plate or four-to-five grooves and ridges/plate. Pits, grooves and ridges were either regular (32 mm width, depth/height and spacing) or variable (8–56 mm). Plates without pits, grooves or ridges were granite fragments set in cement. Granite may be considered an environmentally-sensitive material compared with concrete (see "*Use environmentally-sensitive material on intertidal artificial structures*"). Five of each design were randomly arranged at lowshore on each of two granite boulder seawalls in July 2011. Macroalgae on plates were counted from photographs after 12 months.

A replicated, randomized, controlled study in 2009–2010 on two intertidal seawalls on island coastlines in the Singapore Strait, Singapore (8a; same experimental set-up as 8b) found that concrete settlement plates with pit habitats supported different macroalgae and invertebrate community composition with higher species richness but similar abundances compared with granite plates without pits. After 13 months, macroalgae and invertebrate species richness was higher on settlement plates with pits (11 species/plate) than without (3/plate), while abundances were statistically similar (231 vs 178 individuals/plate). Community composition differed on plates with and without pits (data reported as statistical model results). Settlement plates (200 × 200 mm) were moulded with and without pit habitats. Plates with pits were concrete with 36 square pits/plate with either regular (16 mm width, depth and spacing) or variable (4–28 mm) arrangement. Plates without pits were granite fragments set in cement. Granite may be considered an environmentally-sensitive material compared with concrete (see "*Use environmentally-sensitive material on intertidal artificial structures*"). Eight of each design were randomly arranged at both lowshore and highshore on each of two granite boulder seawalls in November–December 2009. Macroalgae on plates were counted from photographs and invertebrates in the laboratory after 13 months.

A replicated, randomized, controlled study in 2009–2010 on two intertidal seawalls on island coastlines in the Singapore Strait, Singapore (8b; same experimental set-up as 8a) found that concrete settlement plates with pit habitats, along with grooves, small protrusions and small ridges, supported higher macroalgae and invertebrate species richness than granite plates without added habitats, but that abundances and community composition varied depending on the habitat arrangement, shore level and site. After 13 months, macroalgae and invertebrate species richness was higher on settlement plates with pits, grooves, protrusions and ridges than on plates without at lowshore (13–23 vs 6–10 species/plate) and highshore (5–9 vs 2–3/plate). Richness was higher on plates with variable habitats than regular ones at lowshore (22–23 vs 13–16/plate), but not highshore (6–9 vs 5–6/plate). Abundances were higher on plates with added habitats than without in four of eight comparisons (9–833 vs 3–208 individuals/plates), while community composition differed in three of four comparisons (data reported as statistical model results). In all other comparisons, results were similar (abundances: 104–1,957 vs 49–1,162/plate). It is not clear whether these effects were the direct result of creating pits, grooves, protrusions or ridges. However, richness was higher on plate quarters with pits (11 species/quarter) than ridges (6/quarter), but similar to quarters with protrusions and with grooves and ridges (both 8/quarter). Abundances were similar for all four habitats types (88–231 individuals/quarter). Settlement plates (400 × 400 mm) were moulded with and without pit habitats, along with grooves, small protrusions and small ridges. Plates with added habitats were concrete. Each 200 × 200 mm quarter contained either 36 square pits, four-to-five grooves and ridges, 36 protrusions or 12 ridges. All habitats had either regular (16 mm width, depth/height and spacing) or variable (4–28 mm) arrangement. Plates without added habitats were granite fragments set in cement. Granite may be considered an environmentally-sensitive material compared with concrete (see "*Use environmentally-sensitive material on intertidal artificial structures*"). Eight of each design were randomly arranged at both lowshore and highshore on each of two granite boulder seawalls in November–December 2009. Macroalgae on plates were counted from photographs and invertebrates in the laboratory after 13 months.

A replicated, randomized, controlled study in 2006–2014 on an intertidal seawall on open coastline in the Atlantic Ocean, Azores (9; same experimental set-up as 2) found that creating pit habitats on the seawall had mixed effects on macroalgae and invertebrate abundances depending on the species, site and pit size and density. After seven years, abundance was higher on seawall surfaces with pits than those without for limpets *Patella candei* in three of four comparisons (1–20 vs 2 individuals/surface), for barnacles *Chthamalus stellatus* in two of four comparisons (8–27 vs 11% cover), and for periwinkles *Tectarius striatus* in one of four comparisons (2–11 vs 1 individuals/surface). Limpets and barnacles were more abundant on surfaces with large pits (limpets: 8–20/surface; barnacles: 25–27%) than small (limpets: 1–8/surface; barnacles: 8–12%). The opposite was true for periwinkles (large pits: 2/surface; small: 7–11/surface). Limpets were more abundant on surfaces with high-density pits (8–20/surface) than low-density (1–8/surface), whereas abundance did not significantly differ for barnacles (high-density: 12–27%; low: 8–25%) or periwinkles (high: 2–7/surface; low: 2–11/surface). Results were variable for small periwinkles *Melarhaphe neritoides* and macroalgae (see paper for results). Pit habitats were created by drilling into a basalt boulder seawall. Arrays of large (diameter: 24 mm) and small (12 mm) round pits (depth: 10 mm) were evenly-spaced on 250 × 250 mm seawall surfaces with high (16 pits/array) or low (8/array) densities. There were five surfaces with each size-density combination and five without pits, randomly arranged at midshore in each of two sites along the seawall. Limpets were removed from surfaces when pits were created in November 2006, then macroalgae and invertebrates were counted on surfaces with and without pits during low tide after 87 months.

A replicated, randomized, controlled study in 2011–2012 on two intertidal seawalls on island coastlines in the Singapore Strait, Singapore (10; same experimental set-up as 7) found that concrete settlement plates with pit habitats supported higher macroalgae and invertebrate species richness and different community composition compared with granite plates without pits. After 12 months, settlement plates with variable pits supported a total of 49 macroalgae and invertebrate species, while plates with regular pits supported 35 species and plates without pits supported 22 (data not statistically tested). Average richness was similar on plates with variable (17 species/plate) and regular (14/plate) pits, and higher on both than on plates without pits (7/plate). Community composition was similar on plates with variable and regular pits, but both differed to plates without pits (data reported as statistical model results). Settlement plates (400 × 400 mm) were moulded with and without pit habitats. Plates with pits were concrete with 36 square pits/plate with either regular (32 mm width, depth and spacing) or variable (8–56 mm) arrangement. Plates without pits were granite fragments set in cement. Granite may be considered an environmentally-sensitive material compared with concrete (see "*Use environmentally-sensitive material on intertidal artificial structures*"). Five of each design were randomly arranged at lowshore on each of two granite boulder seawalls in July 2011. Macroalgae and invertebrates on plates were counted in the laboratory after 12 months.

A replicated, controlled, before-and-after study in 2014–2015 on an intertidal seawall on open coastline in the North Sea, UK (11a) found that creating pit habitats on the seawall altered the macroalgae and invertebrate community composition and increased their species diversity, richness and abundance on seawall surfaces. After 12 months, the macroalgae and invertebrate species diversity (data reported as Shannon index), richness and abundance were higher on surfaces with pits (3 species/surface, 99 individuals/surface) than without (1 species/surface, 26 individuals/surface), and also compared with surfaces before pits were created (0 species and individuals/surface). Community composition differed on surfaces with and without pits (data reported as statistical model results). One macroalgal species recorded on surfaces with pits was absent from those without. Pit habitats were created by drilling into vertical surfaces of a granite boulder seawall. Round pits were in arrays of four (diameter: 16 mm; depth: 20 mm; 70 mm apart) on 200 × 200 mm seawall surfaces. There were 16 surfaces with pits and 16 without at mid-lowshore. Pits were angled to retain water. Organisms were removed from surfaces when pits were created in October 2014, then macroalgae and invertebrates were counted on surfaces with and without pits during low tide over 12 months.

A replicated, controlled, before-and-after study in 2015–2016 on two intertidal groynes on open coastline in the English Channel, UK (11b) found that creating pit habitats on the groynes altered the macroalgae, invertebrate and fish community composition and increased their species diversity and richness but not abundance on groyne surfaces. After 12 months, macroalgae, invertebrate and fish species diversity (data reported as Shannon index) was higher on surfaces with pits than without, and also compared with surfaces before pits were created. Species richness on surfaces with pits

(2 species/surface) was statistically similar to surfaces without (1/surface), but higher than before pits were created (1/surface). Abundances were lower on surfaces with pits (7 individuals/surface) than without (65/surface), and statistically similar to before pits were created (33/surface). Community composition differed on surfaces with and without pits (data reported as statistical model results). Five species (2 mobile invertebrates, 2 non-mobile invertebrates, 1 fish) recorded on surfaces with pits were absent from those without. Pit habitats were created by drilling into vertical surfaces of two limestone boulder groynes. Round pits were in arrays of four (diameter: 16 mm; depth: 20 mm; 70 mm apart) on 200 × 200 mm groyne surfaces. There were 48 surfaces with pits and 48 without at lowshore. Pits were angled to retain water. Organisms were removed from surfaces when pits were created in March 2015, then macroalgae, invertebrates and fishes were counted on surfaces with and without pits during low tide over 12 months.

A replicated, randomized, controlled study in 2013–2016 on an intertidal seawall on open coastline in the Atlantic Ocean, Azores (12) reported that pit habitats created on the seawall supported more limpets *Patella candei*, periwinkles *Tectarius striatus* and small periwinkles *Melarhaphe neritoides* than seawall surfaces without pits, and found that their species richness and diversity (but not abundance) varied depending on the pit patchiness. After 30 months, average limpet and periwinkle species richness was 2 species/surface with pits and 1/surface without, while average abundances were 2–22 individuals/surface with pits and 0/surface without (data not statistically tested). Species diversity and richness varied depending on the pit patchiness, but average abundances did not. Pit occupancy and the effects of patchiness on total abundances varied by species (see paper for details). Pit habitats were created in December 2013 by drilling into a basalt boulder seawall. Arrays of 16 round pits (diameter: 12 mm; depth: 10 mm) on 250 × 250 mm surfaces had three levels of patchiness: high (4 patches of 4); moderate (2 patches of 8); and low (evenly-spaced). There were five surfaces of each and five without pits, randomly arranged at midshore in each of two sites along the seawall. Seasnails were counted on surfaces with and without pits during low tide after 30 months.

A replicated, randomized, controlled study in 2014–2015 on an intertidal seawall on an island coastline in the Singapore Strait, Singapore (13) found that creating pit habitats on the seawalls, along with grooves, increased the macroalgae and invertebrate species richness on seawall surfaces, and that increasing the density and fragmentation of pits and grooves had mixed effects on species richness. After 12 months, macroalgae and invertebrate species richness was higher on seawall surfaces with pits and grooves (13–29 species/surface) than on surfaces without (3/surface). Species richness varied on surfaces with high-density (19–29/surface), medium-density (14–27/surface) and low-density (13–16/surface) pits and grooves, depending on their arrangement, and *vice versa* (unfragmented arrangement: 14–20/surface; moderately-fragmented: 13–29/surface; highly-fragmented: 15–20/surface). It is not clear whether these effects were the direct result of creating pits or grooves. Concrete settlement plates (200 × 200 mm) were moulded with 37 round pit habitats amongst seven grooves, both with variable length, width and depth (2–56 mm). Plates with pits and grooves were attached to 2.4 × 2.4 m seawall surfaces in varying densities (high: 30 plates/surface; medium: 20/surface; low: 10/surface) and arrangement (unfragmented, moderately-fragmented, highly-fragmented). Four surfaces with each density-fragmentation combination and four with no plates were randomly arranged, spanning low-highshore, on a granite boulder seawall in February 2014. Macroalgae on seawall surfaces with and without plates were counted from photographs and invertebrates in the laboratory after 12 months.

A replicated, randomized, controlled study in 2016–2017 on two intertidal seawalls in the Clyde estuary and on open coastline in the English Channel, UK (14a) found that creating pit habitats on seawall surfaces, along with using environmentally-sensitive material, increased the macroalgae and invertebrate species richness on surfaces at one of two sites, and increased invertebrate abundances at both sites. After 18 months, at one of two sites, macroalgae and mobile invertebrate species richness was higher on settlement plates with pits (2 species/plate) than without (1/plate), but was statistically similar on plates with and without pits at the second site (2 vs 1/plate). At both sites, plates with pits had higher mobile invertebrate abundance (4–11 individuals/plate) and barnacle (Cirripedia) cover (49–74%) than plates without (mobiles: 1/plate; barnacles: 22–34%). It is not clear whether these effects were the direct result of creating pits or using environmentally-sensitive material. Settlement plates (150 × 150 mm) were moulded with and without pit habitats. Plates with pits had multiple irregular pits (maximum depth: 30 mm). Eight limestone-cement (environmentally-sensitive material) plates with pits and eight concrete plates without were randomly arranged at upper-midshore on each of two vertical concrete seawalls in April–May 2016. Macroalgae and invertebrates on plates were counted from photographs over 18 months.

A replicated, randomized, controlled study in 2016–2017 on two intertidal seawalls in the Clyde and Forth estuaries, UK (14b) found that creating pit habitats on the seawalls, along with grooves, had mixed effects on the macroalgae and invertebrate species richness and invertebrate abundances, depending on the site. After 18 months, at one of two sites, macroalgae and mobile invertebrate species richness and mobile invertebrate abundances were higher on settlement plates with pits and grooves (4 species/plate, 11 individuals/plate) than without (1 species/plate, 1 individual/plate), but barnacle (Cirripedia) cover was similar on plates with and without pits and grooves (15 vs 22%). At the second site, richness and mobile invertebrate abundances were similar on plates with and without pits and grooves (2 vs 1 species/plate, both 3 individuals/plate), while barnacle cover was lower on plates with pits and grooves (73 v 83%). It is not clear whether these effects were the direct result of creating pits or grooves. Concrete settlement plates (150 × 150 mm) were moulded with and without pit habitats and grooves. Plates with pits and grooves had 37 round pits amongst seven grooves, both with variable dimensions (maximum depth: 30 mm). Eight plates with pits and grooves and eight without were randomly arranged at upper-midshore on each of two vertical concrete seawalls in April–May 2016. Macroalgae and invertebrates on plates were counted from photographs over 18 months.

A replicated study in 2018–2019 on four intertidal seawalls on island coastlines in the Singapore Strait, Singapore, and in the Plym and Tamar estuaries, UK (15) reported that settlement plates with pit habitats supported macroalgae and invertebrates. Over 12 months, settlement plates with pits supported 67 invertebrate species in total (Singapore: 54; UK: 13). After 12 months, there were 3–21 species/plate and 6–216 individuals/plate. Plates supported 1–5% cover of limpets (Patellidae, Fissurellidae, Siphonariidae, Lottioidea), 18–25% cover of barnacles (Cirripedia), 4–8% cover of ephemeral green macroalgae, and 29–35% cover of encrusting macroalgae. Concrete settlement plates (200 × 200 mm) were moulded with 15 water-retaining round pit habitats (diameter: 6–28 mm; depth not reported) over half their surfaces. Plates had either reduced pH (environmentally-sensitive material) or standard pH. Twenty-four of each were attached at a 60° angle at midshore on each of two seawalls in both Singapore and the UK during February–March 2018. Macroalgae on plates were counted from photographs and invertebrates in the laboratory over 12 months. Eight plates were missing and no longer provided habitat.

A replicated, randomized, controlled study in 2018–2019 on an intertidal seawall on an island coastline in the Singapore Strait, Singapore (16) found that creating pit habitats on the seawall, along with grooves, increased the macroalgae and non-mobile invertebrate abundance, fish species richness and abundance, and altered the fish community composition and behaviour on and around seawall surfaces. After 12 months, macroalgae and non-mobile invertebrate abundance was higher on seawall surfaces with pits and grooves (17% cover) than on surfaces without (4%). Over 12 months, fish community composition differed on and around surfaces with and without pits and grooves (data reported as statistical model results). Fish species richness and maximum abundance were higher on and around surfaces with pits and grooves (9–15 species and 14–29 individuals/60-minute survey) than without (7–14 species/survey, 10–25 individuals/survey), and fishes took more bites from surfaces with pits and grooves (18–456 vs 4–17 bites/survey). Eleven fish species recorded on and around surfaces with pits and grooves were absent from those without. It is not clear whether these effects were the direct result of creating pits or grooves. Concrete settlement plates (200 × 200 mm) were moulded with 37 round pit habitats amongst seven grooves, both with variable length, width and depth (2–56 mm). Twenty plates with pits and grooves were attached to 2.4 × 2.4 m seawall surfaces in seven irregularly-spaced patches. Plates had been naturally-colonized since February 2015. Six surfaces with plates and six without were randomly arranged, spanning low-highshore, on a granite boulder seawall in February 2018. Macroalgae and non-mobile invertebrates on seawall surfaces with and without plates were counted from photographs, while fishes and the number of bites they took were counted from 60-minute videos during each of seven high tides over 12 months.

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2.5. Create hole habitats (>50 mm) on intertidal artificial structures

- **Five studies** examined the effects of creating hole habitats on intertidal artificial structures on the biodiversity of those structures. Three studies were in estuaries in southeast Australia[1,2] and the UK[3], one was on an open coastline in the Netherlands[4], and one was in a marina in northern Israel[5].

COMMUNITY RESPONSE (3 STUDIES)

- **Overall community composition (3 studies):** One replicated, randomized, paired sites, controlled, before-and-after study in Israel[5] found that creating hole habitats on an intertidal artificial structure, along with grooves, small ridges and environmentally-sensitive material, altered the combined macroalgae and invertebrate community composition on structure surfaces. The study, along with two other replicated, controlled studies in Australia[1] and the UK[3], also reported that hole habitats, along with rock pools[1,3], or grooves, small protrusions and environmentally-sensitive material[5], supported macroalgae[1,5] and/or non-mobile invertebrate[1,3,5] species that were absent from structure surfaces without added habitat features.

- **Overall richness/diversity (3 studies):** Three replicated, controlled studies (including one randomized, paired sites, before-and-after study) in Australia[1], the UK[3] and Israel[5] found that creating hole habitats on intertidal artificial structures, along with rock pools[1,3], or grooves, small protrusions and environmentally-sensitive material[5], increased the combined macroalgae and invertebrate species diversity[5] and/or richness[1,3,5] on structure surfaces.

POPULATION RESPONSE (2 STUDIES)

- **Algal abundance (1 study):** One replicated, paired sites, controlled study in the Netherlands[4] reported that creating hole habitats on an intertidal artificial structure did not increase the macroalgal abundance on structure surfaces.
- **Invertebrate abundance (2 studies):** One of two replicated, controlled studies (including one paired sites study) in Australia[1] and the Netherlands[4] reported that creating hole habitats on an intertidal artificial structure did not increase the invertebrate abundance on structure surfaces[4]. One study[1] found that creating holes, along with rock pools, had mixed effects on the limpet abundance, depending on the shore level and site.

BEHAVIOUR (1 STUDY)

- **Use (1 study):** One study in Australia[2] reported that hole habitats created on an intertidal artificial structure, along with rock pools, were used by sea slugs, urchins and octopuses.

Background

Hole habitats provide organisms refuge from desiccation and temperature fluctuations during low tide in intertidal rocky habitats (Williams & Morrit 1995). They also provide shelter from predation or grazing (Menge & Lubchenco 1981). The size and density of holes is likely to affect the size, abundance and variety of organisms that can use them. Small holes can provide refuge for small-bodied organisms but may exclude larger organisms, limit their growth and get rapidly filled-up (Firth *et al.* 2020). Large holes can be used by larger-bodied organisms but may not provide sufficient refuge from predators for smaller organisms. By default, holes contain shaded surfaces, which can be associated with the presence of non-native species (Dafforn 2017).

Holes are sometimes present on boulders used in marine artificial structures as a result of quarrying processes or engineering tests (Firth *et al*. 2014). However, these are sparse when present and are normally absent from other types of structures. Holes sometimes form on artificial structures through erosion, but are often filled or repaired during maintenance works (Moreira *et al.* 2007). Hole habitats can be created on intertidal artificial structures by adding or removing material, either during construction or retrospectively.

Definition: 'Hole habitats' are depressions with a length to width ratio ≤3:1 and depth >50 mm (modified from "Subtidal holes" in Strain *et al.* 2018). Intertidal hole habitats do not retain water during low tide – those that do would come under the action "*Create 'rock pools' on intertidal artificial structures*". The two actions can be combined (e.g. pools created in holes).

See also: *Create natural rocky reef topography on intertidal artificial structures*; *Create pit habitats (1–50 mm) on intertidal artificial structures*; *Create groove habitats (1–50 mm) on intertidal artificial structures*; *Create crevice habitats (>50 mm) on intertidal artificial structures*; *Create 'rock pools' on intertidal artificial structures*; *Create small adjoining cavities or 'swimthrough' habitats (≤100 mm) on intertidal artificial structures*; *Create large adjoining cavities or 'swimthrough' habitats (>100 mm) on intertidal artificial structures*; *Create groove habitats* and *small protrusions, ridges or ledges (1–50 mm) on intertidal artificial structures*.

Dafforn K.A. (2017) Eco-engineering and management strategies for marine infrastructures to reduce establishment and dispersal of non-indigenous species. *Management of Biological Invasions*, 8, 153–161.

Firth L.B., Airoldi L., Bulleri F., Challinor S., Chee S.-Y., Evans A.J., Hanley M.E., Knights A.M., O'Shaughnessy K., Thompson R.C. & Hawkins S.J. (2020) Greening of grey infrastructure should not be used as a Trojan horse to facilitate coastal development. *Journal of Applied Ecology*, 57, 1762–1768.

Firth L.B., Thompson R.C., Bohn K., Abbiati M., Airoldi L., Bouma T.J., Bozzeda F., Ceccherelli V.U., Colangelo M.A., Evans A., Ferrario F., Hanley M.E., Hinz H., Hoggart S.P.G., Jackson J.E., Moore P., Morgan E.H., Perkol-Finkel S., Skov M.W., Strain E.M., van Belzen J. & Hawkins S.J. (2014) Between a rock and a hard place: environmental and engineering considerations when designing coastal defence structures. *Coastal Engineering*, 87, 122–135.

Menge B.A. & Lubchenco J. (1981) Community organization in temperate and tropical rocky intertidal habitats: prey refuges in relation to consumer pressure gradients. *Ecological Monographs*, 51, 429–450.

Moreira J., Chapman M.G. & Underwood A.J. (2007) Maintenance of chitons on seawalls using crevices on sandstone blocks as habitat in Sydney Harbour, Australia. *Journal of Experimental Marine Biology and Ecology*, 347, 134–143.

Strain E.M.A., Olabarria C., Mayer-Pinto M., Cumbo V., Morris R.L., Bugnot A.B., Dafforn K.A., Heery E., Firth L.B., Brooks P.R. & Bishop M.J. (2018) Eco-engineering urban infrastructure for marine and coastal biodiversity: which interventions have the greatest ecological benefit? *Journal of Applied Ecology*, 55, 426–441.

Williams G.A. & Morritt D. (1995) Habitat partitioning and thermal tolerance in a tropical limpet, *Cellana grata*. *Marine Ecology Progress Series*, 124, 89–103.

A replicated, controlled study in 2006–2007 on an intertidal seawall in Sydney Harbour estuary, Australia (1) reported that hole habitats created on the seawall, along with rock pools, supported higher macroalgae and non-mobile invertebrate species richness than seawall surfaces without holes or pools, and found that limpet *Siphonaria denticulata* abundance varied depending on the shore level and site. After 12–14 months, holes supported 2–31 macroalgae and non-mobile invertebrate species groups/site (highshore: 2–10/site; midshore: 16–18/site; lowshore: 27–31/site), while seawall surfaces without holes or pools supported 2–21/site (highshore: 2–3/site; midshore: 7–11/site; lowshore: 19–21/site) (data not statistically tested). At least 14 species (≥5 macroalgae, ≥9 non-mobile invertebrates) recorded in holes were absent from surfaces without. Limpet abundances varied by shore level and site (see paper for results). It is not clear whether these effects were the direct result of creating holes or rock pools. Hole habitats were created during July–September 2006 by replacing seawall blocks with water-retaining troughs during construction of a vertical sandstone seawall. Six cuboidal holes (length: 600 mm; height/depth: 300 mm) were created at highshore, midshore, and lowshore in each of three sites along the seawall. Hole surfaces were sandstone and concrete. Water pooled to 50 mm in the base of holes but wet surfaces were not surveyed. Holes were compared with six seawall surfaces (length: 600 mm; height: 300 mm) at each shore level and site. Macroalgae and invertebrates were counted in holes and on seawall surfaces during low tide in September 2007.

A study (year not reported) on an intertidal seawall in Sydney Harbour estuary, Australia (2) reported that hole habitats created on the seawall, along with rock pools, were used by mobile invertebrates from at least three species groups. Sea slugs (Opistobranchia), urchins (Echinoidea) and octopuses (Octopoda) were recorded in holes and pools. It is not clear whether these effects were the direct result of creating holes or rock pools. Hole habitats were created, along with rock pools, by replacing seawall blocks with sandbags during maintenance of a vertical sandstone seawall, then removing the sandbags to leave shaded water-retaining depressions in the wall. No other details were reported.

A replicated, controlled study in 2010–2011 on an intertidal seawall in the Teign estuary, UK (3) found that hole habitats created on the seawall, along with rock pools, supported higher macroalgae and invertebrate species richness than seawall surfaces without holes or pools. After 19 months, macroalgae and invertebrate species richness was higher in holes (3 species/hole) than on seawall surfaces without (1/surface). Barnacles (Cirripedia) were recorded only in holes. It is not clear whether these effects were the direct result of creating holes or rock pools. Hole habitats were created in May 2010 by replacing seawall blocks with water-retaining troughs during construction of a vertical sandstone seawall. Fifteen cube-shaped holes (150 × 150 × 150 mm) were created at highshore. Water pooled in the base of holes (depth/volume not reported). Holes were compared with 15 mortar seawall surfaces (150 × 150 mm). Macroalgae and invertebrates were counted in holes and on seawall surfaces during low tide after 19 months. Three holes and seven surfaces had been buried by sediment and no longer provided habitat.

A replicated, paired sites, controlled study in 2008–2010 on an intertidal breakwater on open coastline in the North Sea, Netherlands (4) reported that settlement plates with hole habitats supported similar abundances of macroalgae and invertebrates to plates without holes. Data were not statistically tested. After 28 months, there were no clear differences in macroalgal or invertebrate abundances on plates with and without holes (data not reported). Concrete settlement plates (250 × 250 mm) were made with and without hole habitats using a mould. Plates with holes had one hemispherical hole/plate (diameter: 150 mm; depth: 50 mm). One plate with a hole and one without were placed on each of 10 horizontal surfaces on each side of a concrete-block breakwater (wave-exposed, wave-sheltered) in May 2008. On the wave-exposed side, plates were at mid-highshore, while on the wave-sheltered side, plates were at low-midshore. Macroalgae and invertebrates on plates were counted during low tide over 28 months.

A replicated, randomized, paired sites, controlled, before-and-after study in 2014–2016 on an intertidal seawall in a marina in the Mediterranean Sea, Israel (5) found that hole habitats created on seawall panels, along with grooves, small ledges and environmentally-sensitive material, supported higher macroalgae and invertebrate species diversity and richness and different community composition compared with standard-concrete seawall surfaces without added habitats. After 22 months, macroalgae and invertebrate species diversity (data reported as Shannon index) and richness was higher on panels with added habitats (8 species/quadrat) than on seawall surfaces without (3/quadrat), and compared with seawall surfaces before habitats were added (2/quadrat). Community composition differed between panels with added habitats and seawall surfaces without (data reported as statistical model results). Five species groups (1 macroalgae, 4 non-mobile invertebrates) recorded on panels were absent from surfaces without. It is not clear whether these effects were the direct result of creating holes, grooves, ledges, or using environmentally-sensitive material. Hole habitats were created on seawall panels (height: 1.5 m; width: 0.9 m; thickness: 130 mm) using a formliner. Each panel had six cylindrical holes (diameter: 30 mm; depth: 120 mm; ≥300 mm apart) amongst multiple grooves and small ledges. Panels were made from patented ECOncrete™ material. Four panels were attached to a vertical concrete seawall in November 2014. The top 0.3 m were intertidal. Seawall surfaces were intertidal areas of seawall cleared of organisms (height: 0.3 m; width: 0.9 m) adjacent to each panel. Macroalgae and invertebrates were counted in one 300 × 300 mm randomly-placed quadrat on each panel and seawall surface during high tide over 22 months.

(1) Chapman M.G. & Blockley D.J. (2009) Engineering novel habitats on urban infrastructure to increase intertidal biodiversity. *Oecologia*, 161, 625–635.

(2) Chapman M.G. & Underwood A.J. (2011) Evaluation of ecological engineering of "armoured" shorelines to improve their value as habitat. *Journal of Experimental Marine Biology and Ecology*, 400, 302–313.

(3) Firth L.B., Thompson R.C., Bohn K., Abbiati M., Airoldi L., Bouma T.J., Bozzeda F., Ceccherelli V.U., Colangelo M.A., Evans A., Ferrario F., Hanley M.E., Hinz H., Hoggart S.P.G., Jackson J.E., Moore P., Morgan E.H., Perkol-Finkel S., Skov M.W., Strain E.M., van Belzen J. & Hawkins S.J. (2014) Between a rock and a hard place: environmental and engineering considerations when designing coastal defence structures. *Coastal Engineering*, 87, 122–135.

(4) Paalvast P. (2015) *The role of geometric structure and texture on concrete for algal and macrofaunal colonization in the marine and estuarine intertidal zone.* Proceedings of RECIF Conference on artificial reefs: From materials to ecosystems. Caen, France, 77–84.

(5) Perkol-Finkel S., Hadary T., Rella A., Shirazi R. & Sella I. (2018) Seascape architecture – incorporating ecological considerations in design of coastal and marine infrastructure. *Ecological Engineering*, 120, 645–654.

2.6. Create groove habitats (1–50 mm) on intertidal artificial structures

- **Fourteen studies** examined the effects of creating groove habitats on intertidal artificial structures on the biodiversity of those structures. Seven studies were in estuaries in southeast Australia[1a,1b,2,9], the UK[3a,7] and Hong Kong[8], four were on open coastlines in the UK[3b,5a,5b] and the Netherlands[4a], two were on island coastlines in the Singapore Strait[6,10], and one was in a port in the Netherlands[4b].

COMMUNITY RESPONSE (11 STUDIES)

- **Overall community composition (3 studies):** Two of three replicated, controlled studies (including one randomized and two before-and-after studies) in Australia[1a] and the UK[5a,5b] found that creating groove habitats on intertidal artificial structures did not alter the combined macroalgae and invertebrate community composition on structure surfaces[1a,5b]. However, one of these studies[5b] reported that grooves supported macroalgae, mobile and non-mobile invertebrate species that were absent from structure surfaces without grooves. One study[5a] found that creating grooves did alter the community composition.
- **Fish community composition (1 study):** One replicated, randomized, controlled study in Singapore[10] found that groove habitats created on an intertidal artificial structure, along with pits, altered the fish community composition on and around structure surfaces, and supported species that were absent from surfaces without grooves and pits.
- **Overall richness/diversity (8 studies):** Three of six replicated, controlled studies (including two randomized and two before-and-after studies) in the UK[3a,3b,5a,5b,7] and Singapore[6] found that creating groove habitats on intertidal artificial structures, along with pits in one study[6], increased the combined macroalgae and invertebrate species richness[5a,5b,6] and/or diversity[5a,5b] on structure surfaces. Two studies[3a,3b] found that creating grooves did not increase their species richness. One[7] found that creating grooves, along with pits, had mixed effects on species richness depending on the site. One of the studies[6] found that increasing the density and fragmentation of grooves, along with pits, had mixed effects on species richness. Two replicated studies (including one randomized, paired sites study) in Hong Kong[8] and Australia[9] found that grooves supported higher species richness than small ridges[8] or ledges[9] created in between them, but one[8] found that species diversity in grooves vs ridges varied depending on the groove depth.
- **Algal richness/diversity (1 study):** One replicated, paired sites, controlled study in Australia[2] found that creating groove habitats on intertidal artificial structures did not increase the macroalgal species richness on structure surfaces.
- **Invertebrate richness/diversity (3 studies):** Two replicated, controlled studies (including one randomized and one paired sites study) in Australia[1a,2] found that creating groove habitats on intertidal artificial structures did not increase the species richness of mobile or non-mobile invertebrates[2] or limpets[1a] on structure surfaces. One replicated study in Australia[9] found that grooves supported higher mobile invertebrate species richness than small ledges created in between them.

- **Fish richness/diversity (2 studies):** One replicated, randomized, controlled study in Singapore[10] found that creating groove habitats on an intertidal artificial structure, along with pits, increased the fish species richness on and around structure surfaces. One replicated study in Australia[9] found that grooves supported similar fish species richness to small ledges created in between them.

POPULATION RESPONSE (9 STUDIES)

- **Overall abundance (4 studies):** Two of three replicated, controlled studies (including one randomized and two before-and-after studies) in the UK[5a,5b] and Singapore[10] found that creating groove habitats on intertidal artificial structures, along with pits in one study[10], increased the combined macroalgae and invertebrate abundance on structure surfaces[5a,10]. One[5b] found that creating grooves did not increase their abundance. One replicated study in Australia[9] found that grooves supported similar abundances to small ledges created in between them.
- **Algal abundance (2 studies):** Two replicated, paired sites, controlled studies in the Netherlands[4a,4b] reported that creating groove habitats on intertidal artificial structures did not increase the macroalgal abundance on structure surfaces.
- **Invertebrate abundance (6 studies):** Three of four replicated, controlled studies (including two randomized and two paired sites studies) in Australia[1a], the Netherlands[4a,4b] and the UK[7] found that creating groove habitats on intertidal artificial structures did not increase the invertebrate[4a,4b], limpet[1a] or chiton[1a] abundances on structure surfaces. One study[7] found that creating grooves, along with pits, had mixed effects on mobile invertebrate and barnacle abundances, depending on the site. One replicated, paired sites, controlled study in Australia[1b] reported that grooves supported non-mobile invertebrates more frequently than structure surfaces without grooves, but not mobile invertebrates. One replicated study in Australia[9] found that grooves supported higher mobile invertebrate and oyster abundances than small ledges created in between them.
- **Fish abundance (2 studies):** One replicated, randomized, controlled study in Singapore[10] found that creating groove habitats on an intertidal artificial structure, along with pits, increased the fish abundance on and around structure surfaces. One replicated study in Australia[9] found that grooves supported similar fish abundance to small ledges created in between them.

BEHAVIOUR (2 STUDIES)

- **Use (1 study):** One replicated, paired sites, controlled study in the Netherlands[4a] reported that groove habitats created on an intertidal artificial structure were used by mussels and periwinkles.
- **Fish behaviour change (1 study):** One replicated, randomized, controlled study in Singapore[10] found that creating groove habitats on an intertidal artificial structure, along with pits, increased the number of bites fishes took from structure surfaces.

Background

Groove habitats provide organisms refuge from desiccation and temperature fluctuations during low tide in intertidal rocky habitats (Williams & Morrit 1995). They also provide shelter from predation or grazing (Menge & Lubchenco 1981) and some species preferentially settle into them (Chabot & Bourget 1988). The size and density of grooves is likely to affect the size, abundance and variety of organisms that can use them. Small grooves can provide refuge for small-bodied organisms but may exclude larger organisms, limit their growth and get rapidly filled-up (Firth *et al.* 2020). Large grooves can be used by larger-bodied organisms but may not provide sufficient refuge from predators for smaller organisms.

Grooves are sometimes present on boulders used in marine artificial structures as a result of quarrying processes (MacArthur *et al.* 2020). They can also form on structures through erosion, but will often be filled or repaired during maintenance works (Moreira *et al.* 2007), and are absent from many structures (Aguilera *et al.* 2014). Groove habitats can be created on intertidal artificial structures by adding or removing material, either during construction or retrospectively.

There is a body of literature investigating the effects of creating groove habitats on experimental artificial substrates on the recruitment and survival of intertidal rocky shore species (e.g. Savoya & Schwindt 2010; van Tamelen *et al.* 1997). These studies are not included in this synopsis, which focusses on *in situ* conservation actions to enhance the biodiversity of marine artificial structures.

Definition: 'Groove habitats' are depressions with a length to width ratio >3:1 and depth 1–50 mm (modified from "Crevices" in Strain *et al.* 2018).

See also: *Create textured surfaces (≤1 mm) on intertidal artificial structures*; *Create natural rocky reef topography on intertidal artificial structures*; *Create pit habitats (1–50 mm) on intertidal artificial structures*; *Create hole habitats (>50 mm) on intertidal artificial structures*; *Create crevice habitats (>50 mm) on intertidal artificial structures*; *Create groove habitats* and *small protrusions, ridges or ledges (1–50 mm) on intertidal artificial structures*.

Aguilera M.A., Broitman B.R. & Thiel M. (2014) Spatial variability in community composition on a granite breakwater versus natural rocky shores: lack of microhabitats suppresses intertidal biodiversity. *Marine Pollution Bulletin*, 87, 257–268.

Chabot R. & Bourget E. (1988) Influence of substratum heterogeneity and settled barnacle density on the settlement of cypris larvae. *Marine Biology*, 97, 45–56.

Firth L.B., Airoldi L., Bulleri F., Challinor S., Chee S.-Y., Evans A.J., Hanley M.E., Knights A.M., O'Shaughnessy K., Thompson R.C. & Hawkins S.J. (2020) Greening of grey infrastructure should not be used as a Trojan horse to facilitate coastal development. *Journal of Applied Ecology*, 57, 1762–1768.

MacArthur M., Naylor L.A., Hansom J.D. & Burrows M.T. (2020) Ecological enhancement of coastal engineering structures: passive enhancement techniques. *Science of the Total Environment*, 740, 139981.

Menge B.A. & Lubchenco J. (1981) Community organization in temperate and tropical rocky intertidal habitats: prey refuges in relation to consumer pressure gradients. *Ecological Monographs*, 51, 429–450.

Moreira J., Chapman M.G. & Underwood A.J. (2007) Maintenance of chitons on seawalls using crevices on sandstone blocks as habitat in Sydney Harbour, Australia. *Journal of Experimental Marine Biology and Ecology*, 347, 134–143.

Savoya V. & Schwindt E. (2010) Effect of the substratum in the recruitment and survival of the introduced barnacle *Balanus glandula* (Darwin 1854) in Patagonia, Argentina. *Journal of Experimental Marine Biology and Ecology*, 382, 125–130.

Strain E.M.A., Olabarria C., Mayer-Pinto M., Cumbo V., Morris R.L., Bugnot A.B., Dafforn K.A., Heery E., Firth L.B., Brooks P.R. & Bishop M.J. (2018) Eco-engineering urban infrastructure for marine and coastal biodiversity: which interventions have the greatest ecological benefit? *Journal of Applied Ecology*, 55, 426–441.

van Tamelen P.G., Stekoll M.S. & Deysher L. (1997) Recovery processes of the brown alga *Fucus gardneri* following the 'Exxon Valdez' oil spill: settlement and recruitment. *Marine Ecology Progress Series*, 160, 265–277.

Williams G.A. & Morritt D. (1995) Habitat partitioning and thermal tolerance in a tropical limpet, *Cellana grata*. *Marine Ecology Progress Series*, 124, 89–103.

A replicated, randomized, controlled study in 2000–2003 on an intertidal seawall in Sydney Harbour estuary, Australia (1a) found that creating groove habitats on the seawall did not alter the macroalgae and invertebrate community composition or increase limpet (Patellidae and/or Siphonariidae, Fissurellidae) species richness or abundance, or chiton (Polyplacophora) abundance on seawall surfaces. After three months, seawall surfaces with grooves supported similar macroalgae and invertebrate community composition to surfaces without (data reported as statistical model results). After 27 months, limpet species richness and abundance were similar in grooves (0 species and individuals/array) and on surfaces without (1 species/surface, 3 individuals/surface). The same was true for chiton abundance (grooves: 0 individuals/array; surfaces: 0/surface). Groove habitats were created in 2000 (month not reported) by drilling into a vertical sandstone seawall during reconstruction. Arrays of 16 grooves (length: 50 mm; width: 10 mm; depth: 5 mm; spacing/orientation not reported) were drilled on 1 × 0.4 m seawall surfaces. There were five surfaces with grooves and five without randomly arranged (shore level not reported). Macroalgae and invertebrates were counted on surfaces with and without grooves during low tide after three months. Mobile invertebrates were counted in grooves and on surfaces without after 27 months.

A replicated, paired sites, controlled study (year not reported) on two intertidal seawalls in Sydney Harbour estuary, Australia (1b; same experimental set-up as 2) reported that creating groove habitats on the seawalls had mixed effects on invertebrate abundances depending on the species group and shore level, but data were not statistically tested. Over 12 months, both grooves and seawall surfaces without grooves supported mobile invertebrates (data not reported) and non-mobile invertebrates at midshore (mussels *Mytilus galloprovincialis planulatis*: recorded in grooves in 65% of surveys vs seawall surfaces in 27%; sponges (Porifera): 10 vs 4%; barnacles (Cirripedia): 26 vs 16%; tubeworms (Polychaeta): 34 vs 26%) and lowshore (mussels: 50 vs 33%; sponges: 20 vs 13%; sea squirts (Ascidiacea): 4 vs 3%; tubeworms: 44 vs 49%). Groove habitats were created by indenting wet mortar between blocks during maintenance of vertical sandstone seawalls (month/year not reported). Five grooves (width: 30–50 mm; depth: 20 mm; length/orientation/spacing not reported) were compared with five flat mortar surfaces (dimensions not reported) at both midshore and lowshore in each of two paired sites on each of two seawalls. Invertebrates were counted in grooves and on surfaces without during low tide over 12 months, on 10 occasions on one seawall and seven on the other.

A replicated, paired sites, controlled study (year not reported) on two intertidal seawalls in Sydney Harbour estuary, Australia (2; same experimental set-up as 1b) found that groove habitats created on the seawalls supported similar macroalgae and invertebrate species richness to seawall surfaces without grooves. Over 12 months, macroalgal species richness was similar in grooves and on surfaces without at lowshore (grooves: 7–8 species/survey; surfaces: 6–9/survey) and midshore (7–8 vs 5–7/survey). The same was true for mobile invertebrates (lowshore: grooves and surfaces both 0–1/survey; midshore: both 2–3/survey) and non-mobile invertebrates (lowshore: 4–6 vs 3–5/survey; midshore: 5–6 vs 3–5/survey). Groove habitats were created by indenting wet mortar between blocks during maintenance of vertical sandstone seawalls (month/year not reported). Five grooves (width: 30–50 mm; depth: 20 mm; length/orientation/spacing not reported) were compared with five flat mortar surfaces (dimensions not reported) at both midshore and lowshore in each of two paired sites on each of two seawalls. Macroalgae and invertebrates were counted in grooves and on surfaces without during low tide over 12 months, on nine occasions on one seawall and seven on the other. Method details reported from 1b.

A replicated, controlled study in 2010–2011 on an intertidal seawall in the Teign estuary, UK (3a) found that groove habitats created on the seawall supported similar macroalgae and invertebrate species richness to seawall surfaces without grooves. After 19 months, macroalgae and invertebrate species richness was similar in grooves (1 species/array) and on surfaces without grooves (1/surface). Groove habitats were created in May 2010 by scraping a trowel across wet mortar between blocks during construction of a vertical sandstone seawall. Arrays of 5–10 grooves (length: 150 mm; width/depth: 1–5 mm) were irregularly-spaced on 150 × 150 mm seawall surfaces. There were 15 surfaces with grooves and 15 without at highshore. Macroalgae and invertebrates were counted in grooves and on surfaces without during low tide after 19 months. Two arrays of grooves and seven surfaces without had been buried by sediment and no longer provided habitat.

A replicated, controlled study in 2012–2013 on an intertidal groyne on open coastline in the Irish Sea, UK (3b) reported that groove habitats created on a concrete block placed in the groyne supported similar macroalgae and invertebrate species richness to groyne surfaces without grooves. Data were not statistically tested. After 13 months, a total of four species were recorded in grooves and on groyne surfaces without grooves. Groove habitats were created on two vertical sides of a concrete block (1.5 × 1.5 × 1 m) using a mould. Ten horizontal grooves (length: 1 m; width/depth: 50 mm) were cast 50 mm apart on each side. The block was placed at midshore in a boulder groyne during construction in February 2012. Surfaces with grooves were compared with vertical surfaces of adjacent groyne boulders (dimensions/material not reported). Macroalgae and invertebrates were counted in grooves and on groyne surfaces without during low tide after 13 months.

A replicated, paired sites, controlled study in 2008–2010 on an intertidal breakwater on open coastline in the North Sea, Netherlands (4a) reported that settlement plates with groove habitats supported similar abundances of macroalgae and invertebrates to plates without grooves. Data were not statistically tested. After 28 months, there were no clear differences in macroalgal or invertebrate abundances on plates with and without grooves (data not reported). Blue mussels *Mytilus edulis* and periwinkles *Littorina saxatilis* and *Littorina neritoides* were seen using grooves. Concrete settlement plates (250 × 250 mm) were made with and without groove habitats using a mould. Plates with grooves had five variable grooves/plate (length: 250 mm; width: 10–35 mm; depth: 20–40 mm) in horizontal or vertical orientation. One of each orientation and one plate without grooves were placed on each of 10 horizontal and 10 vertical surfaces on each side of a concrete-block breakwater (wave-exposed, wave-sheltered) in May 2008. On the wave-exposed side, plates were at mid-highshore, while on the wave-sheltered side, plates were at low-midshore. Macroalgae and invertebrates on plates were counted during low tide over 28 months.

A replicated, paired sites, controlled study in 2009 on 14 jetty pilings in Rotterdam Port in the Rhine-Meuse estuary, Netherlands (4b) reported that settlement plates with groove habitats supported similar abundances of macroalgae and invertebrates to plates without grooves. Data were not statistically tested. After nine months, there were no clear differences in macroalgal or invertebrate abundances on plates with and without grooves (data not reported). Concrete settlement plates (250 × 250 mm) were made with and without groove habitats using a mould. Plates with grooves had five variable grooves/plate (length: 250 mm; width: 10–35 mm; depth: 20–40 mm) in horizontal orientation. One plate with grooves and one without were attached to vertical surfaces on each of 14 wooden pilings at lowshore in March 2009. Macroalgae and invertebrates on plates were counted during low tide over nine months.

A replicated, controlled, before-and-after study in 2014–2015 on an intertidal seawall on open coastline in the North Sea, UK (5a) found that creating groove habitats on the seawall altered the macroalgae and invertebrate community composition and increased their species diversity, richness and abundance on seawall surfaces. After 12 months, the macroalgae and invertebrate species diversity (data reported as Shannon index) was similar on seawall surfaces with and without grooves, but higher than on surfaces before grooves were created. Species richness and abundance were higher on surfaces with grooves (5 species/surface, 183 individuals/surface) than without (2 species/surface, 12 individuals/surface), and also compared with before grooves were created (0 species and individuals/surface). Community composition differed on surfaces with and without grooves (data reported as statistical model results). Groove habitats were created by cutting into vertical surfaces of a granite boulder seawall. Arrays of nine horizontal grooves (length: 600 mm; width: 3–20 mm; depth: 10 mm) were irregularly-spaced on 600 × 600 mm seawall surfaces. There were seven surfaces with grooves and seven without at mid-lowshore. Organisms were removed from surfaces when grooves were created in October 2014, then macroalgae and invertebrates were counted on surfaces with and without grooves during low tide over 12 months.

A replicated, controlled, before-and-after study in 2015–2016 on two intertidal groynes on open coastline in the English Channel, UK (5b) found that creating groove habitats on the groynes increased the macroalgae and invertebrate species diversity and richness on groyne surfaces, but did not increase their abundance or alter the community composition. After 12 months, the macroalgae and invertebrate species diversity (data reported as Shannon index) was similar on groyne surfaces with and without grooves, but higher than on surfaces before grooves were created. Species richness was higher on surfaces with grooves (5 species/surface) than without (2/surface), and also compared with before grooves were created (1/surface). Abundances were similar on surfaces with grooves (55 individuals/surface) and without (75/surface) to surfaces before grooves were created (71/surface). Twelve species (5 macroalgae, 4 mobile invertebrates, 3 non-mobile invertebrates) recorded on surfaces with grooves were absent from those without, but the community composition was similar (data reported as statistical model results). Groove habitats were created by cutting into vertical surfaces of two limestone boulder groynes. Arrays of nine horizontal grooves (length: 600 mm; width: 3–20 mm; depth: 10 mm) were irregularly-spaced on 600 × 600 mm groyne surfaces. There were 24 surfaces with grooves and 24 without at lowshore. Organisms were removed from surfaces when grooves were created in March 2015, then macroalgae and invertebrates were counted on surfaces with and without grooves during low tide over 12 months.

A replicated, randomized, controlled study in 2014–2015 on an intertidal seawall on an island coastline in the Singapore Strait, Singapore (6) found that creating groove habitats on the seawalls, along with pits, increased the macroalgae and invertebrate species richness on seawall surfaces, and that increasing the density and fragmentation of grooves and pits had mixed effects on species richness. After 12 months, macroalgae and invertebrate species richness was higher on seawall surfaces with grooves and pits (13–29 species/surface) than on surfaces without (3/surface). Species richness varied on surfaces with high-density (19–29/surface), medium-density (14–27/surface) and low-density (13–16/surface) grooves and pits, depending on their arrangement, and *vice versa* (unfragmented arrangement: 14–20/surface; moderately-fragmented: 13–29/surface; highly-fragmented: 15–20/surface). It is not clear whether these effects were the direct result of creating grooves or pits. Concrete settlement plates (200 × 200 mm) were moulded with seven groove habitats amongst 37 pits, both with variable length, width and depth (2–56 mm). Plates with grooves and pits were attached to 2.4 × 2.4 m seawall surfaces in varying densities (high: 30 plates/surface; medium: 20/surface; low: 10/surface) and arrangement (unfragmented, moderately-fragmented, highly-fragmented). Four surfaces with each density-fragmentation combination and four with no plates were randomly arranged, spanning low-highshore, on a granite boulder seawall in February 2014. Macroalgae on seawall surfaces with and without plates were counted from photographs and invertebrates in the laboratory after 12 months.

A replicated, randomized, controlled study in 2016–2017 on two intertidal seawalls in the Clyde and Forth estuaries, UK (7) found that creating groove habitats on the seawalls, along with pits, had mixed effects on the macroalgae and invertebrate species richness and invertebrate abundances, depending on the site. After 18 months, at one of two sites, macroalgae and mobile invertebrate species richness and mobile invertebrate abundances were higher on settlement plates with grooves and pits (4 species/plate, 11 individuals/plate) than without (1 species/plate, 1 individual/plate), but barnacle (Cirripedia) cover was similar on plates with and without grooves and pits (15 vs 22%). At the second site, richness and mobile invertebrate abundances were similar on plates with and without grooves and pits (2 vs 1 species/plate, both 3 individuals/plate), while barnacle cover was lower on plates with grooves and pits (73 v 83%). It is not clear whether these effects were the direct result of creating grooves or pits. Concrete settlement plates (150 × 150 mm) were moulded with and without groove habitats and pits. Plates with grooves and pits had seven grooves amongst 37 pits, both with variable dimensions (maximum depth: 30 mm). Eight plates with grooves and pits and eight without were randomly arranged at upper-midshore on each of two vertical concrete seawalls in April–May 2016. Macroalgae and invertebrates on plates were counted from photographs over 18 months.

A replicated, randomized, paired sites study in 2016–2017 on two intertidal seawalls in the Pearl River estuary, Hong Kong (8) found that groove habitats created on the seawalls supported higher macroalgae and invertebrate species richness than small ridges created in between them, while species diversity varied depending on the groove depth. After 12 months, macroalgae and invertebrate species richness on settlement plates was similar in deep (8–9 species/plate) and shallow (9/plate) grooves, and higher in both than on the ridges in between (both 3–4/plate). The same was true for species diversity, except that deep grooves supported similar diversity to ridges (data reported as Shannon index). Concrete settlement plates (250 × 250 mm) were moulded with four deep (depth: 50 mm) or shallow (25 mm) vertical groove habitats (length: 250 mm; width: 15–50 mm) between five small ridges (length: 250 mm; width: 17–65 mm; height: 50 or 25 mm). Five of each were randomly arranged at midshore on each of two vertical concrete seawalls in November 2016 (month/year: M. Perkins *pers. comms.*). Plates had textured surfaces. Macroalgae and invertebrates in grooves and on ridges were counted in the laboratory after 12 months. One plate with deep grooves was missing and no longer provided habitat.

A replicated study in 2015–2016 on two intertidal seawalls in Sydney Harbour estuary, Australia (9) found that groove habitats created on the seawalls supported higher macroalgae and invertebrate species richness and higher mobile invertebrate and oyster *Saccostrea glomerata* abundances than small ledges created in between them, but that macroalgae and other non-mobile invertebrate abundance, fish species richness and fish abundance were similar in grooves and on ledges. After 12 months, grooves supported higher macroalgae and non-mobile invertebrate species richness (6 species/groove) than the ledges in between (4/ledge). The same was true for mobile invertebrates (6/groove vs 4/ledge), but not fishes (both 2/sample). Abundances were higher in grooves than on ledges for mobile invertebrates (18 individuals/groove vs 1/ledge) and oysters (56 vs 14% cover), but were similar for macroalgae and other non-mobile invertebrates (38 vs 36% cover) and fishes (both 1 individual/sample). Concrete settlement plates (250 × 250 mm) were moulded with four horizontal groove habitats (length: 250 mm; width: 15–50 mm; depth: 50 mm) between five small ledges (length: 250 mm; width: 17–65 mm; height: 50 mm). Five plates were attached at midshore on each of two vertical sandstone seawalls in November 2015. Plates had textured surfaces. Macroalgae and invertebrates were counted in grooves and on ledges during low tide, from photographs and in the laboratory after 12 months. Fishes were counted from time-lapsed photographs during two high tides.

A replicated, randomized, controlled study in 2018–2019 on an intertidal seawall on an island coastline in the Singapore Strait, Singapore (10) found that creating groove habitats on the seawall, along with pits, increased the macroalgae and non-mobile invertebrate abundance, fish species richness and abundance, and altered the fish community composition and behaviour on and around seawall surfaces. After 12 months, macroalgae and non-mobile invertebrate abundance was higher on seawall surfaces with grooves and pits (17% cover) than on surfaces without (4%). Over 12 months, fish community composition differed on and around surfaces with and without grooves and pits (data reported as statistical model results). Fish species richness and maximum abundance were higher on and around surfaces with grooves and pits (9–15 species and 14–29 individuals/60-minute survey) than without (7–14 species/survey, 10–25 individuals/survey), and fishes took more bites from surfaces with grooves and pits (18–456 vs 4–17 bites/survey). Eleven fish species recorded on and around surfaces with grooves and pits were absent from those without. It is not clear whether these effects were the direct result of creating grooves or pits. Concrete settlement plates (200 × 200 mm) were moulded with seven groove habitats amongst 37 pits, both with variable length, width and depth (2–56 mm). Twenty plates with grooves and pits were attached to 2.4 × 2.4 m seawall surfaces in seven irregularly-spaced patches. Plates had been naturally-colonized since February 2015. Six surfaces with plates and six without were randomly arranged, spanning low-highshore, on a granite boulder seawall in February 2018. Macroalgae and non-mobile invertebrates on seawall surfaces with and without plates were counted from photographs, while fishes and the number of bites they took were counted from 60-minute videos during each of seven high tides over 12 months.

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(2) Dugan J.E., Airoldi L., Chapman M.G., Walker S.J. & Schlacher T. (2011) Estuarine and coastal structures: environmental effects, a focus on shore and nearshore structures. Pages 17–41 in: E. Wolanski & D.S. McLusky (eds.) *Treatise on Estuarine and Coastal Science Vol 8*. Academic Press, Waltham, Massachusetts.

(3) Firth L.B., Thompson R.C., Bohn K., Abbiati M., Airoldi L., Bouma T.J., Bozzeda F., Ceccherelli V.U., Colangelo M.A., Evans A., Ferrario F., Hanley M.E., Hinz H., Hoggart S.P.G., Jackson J.E., Moore P., Morgan E.H., Perkol-Finkel S., Skov M.W., Strain E.M., van Belzen J. & Hawkins S.J. (2014) Between a rock and a hard place: environmental and engineering considerations when designing coastal defence structures. *Coastal Engineering*, 87, 122–135.

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(6) Loke L.H.L., Chisholm R.A. & Todd P.A. (2019) Effects of habitat area and spatial configuration on biodiversity in an experimental intertidal community. *Ecology*, 100, e02757.

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(8) Bradford T.E., Astudillo J.C., Lau E.T.C., Perkins M.J., Lo C.C., Li T.C.H., Lam C.S., Ng T.P.T., Strain E.M.A., Steinberg P.D. & Leung K.M.Y. (2020) Provision of refugia and seeding with native bivalves can enhance biodiversity on vertical seawalls. *Marine Pollution Bulletin*, 160, 111578.
(9) Strain E.M.A., Cumbo V.R., Morris R.L., Steinberg P.D. & Bishop M.J. (2020) Interacting effects of habitat structure and seeding with oysters on the intertidal biodiversity of seawalls. *PLoS ONE*, 15, e0230807.
(10) Taira D., Heery E.C., Loke L.H.L., Teo A., Bauman A.G. & Todd P.A. (2020) Ecological engineering across organismal scales: trophic-mediated positive effects of microhabitat enhancement on fishes. *Marine Ecology Progress Series*, 656, 181–192.

2.7. Create crevice habitats (>50 mm) on intertidal artificial structures

- We found no studies that evaluated the effects of creating crevice habitats on intertidal artificial structures on the biodiversity of those structures.

This means we did not find any studies that directly evaluated this intervention during our literature searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Background

Crevice habitats provide organisms refuge from desiccation and temperature fluctuations during low tide in intertidal rocky habitats (Williams & Morritt 1995). They also provide shelter from predation or grazing (Menge & Lubchenco 1981) and some species preferentially settle into them (Chabot & Bourget 1988). The size and density of crevices is likely to affect the size, abundance and variety of organisms that can use them. Small crevices can provide refuge for small-bodied organisms but may exclude larger organisms, limit their growth and get rapidly filled-up (Firth *et al.* 2020). Large crevices can be used by larger-bodied organisms but may not provide sufficient refuge from predators for smaller organisms. By default, crevices contain shaded surfaces, which can be associated with the presence of non-native species (Dafforn 2017).

Crevices sometimes form on artificial structures through erosion. However, these are often filled or repaired during maintenance works (Moreira *et al.* 2007) and are absent from many structures (Aguilera *et al.* 2014). Crevice habitats can be created on intertidal artificial structures by adding or removing material, either during construction or retrospectively.

Definition: ‘Crevice habitats’ are depressions with a length to width ratio >3:1 and depth >50 mm (modified “Crevices” from Strain *et al.* 2018).

See also: *Create natural rocky reef topography on intertidal artificial structures*; *Create pit habitats (1–50 mm) on intertidal artificial structures*; *Create hole habitats (>50 mm) on intertidal artificial structures*; *Create groove habitats (1–50 mm) on intertidal artificial structures*; *Create ‘rock pools’ on intertidal artificial structures*; *Create small adjoining cavities or ‘swimthrough’ habitats (≤100 mm) on intertidal artificial structures*; *Create* *large adjoining cavities or 'swimthrough' habitats (>100 mm) on intertidal artificial structures*; *Create groove habitats* and *small protrusions, ridges or ledges (1–50 mm) on intertidal artificial structures*.

Aguilera M.A., Broitman B.R. & Thiel M. (2014) Spatial variability in community composition on a granite breakwater versus natural rocky shores: lack of microhabitats suppresses intertidal biodiversity. *Marine Pollution Bulletin*, 87, 257–268.

Chabot R. & Bourget E. (1988) Influence of substratum heterogeneity and settled barnacle density on the settlement of cypris larvae. *Marine Biology*, 97, 45–56.

Dafforn K.A. (2017) Eco-engineering and management strategies for marine infrastructures to reduce establishment and dispersal of non-indigenous species. *Management of Biological Invasions*, 8, 153–161.

Firth L.B., Airoldi L., Bulleri F., Challinor S., Chee S.-Y., Evans A.J., Hanley M.E., Knights A.M., O'Shaughnessy K., Thompson R.C. & Hawkins S.J. (2020) Greening of grey infrastructure should not be used as a Trojan horse to facilitate coastal development. *Journal of Applied Ecology*, 57, 1762–1768.

Menge B.A. & Lubchenco J. (1981) Community organization in temperate and tropical rocky intertidal habitats: prey refuges in relation to consumer pressure gradients. *Ecological Monographs*, 51, 429–450.

Moreira J., Chapman M.G. & Underwood A.J. (2007) Maintenance of chitons on seawalls using crevices on sandstone blocks as habitat in Sydney Harbour, Australia. *Journal of Experimental Marine Biology and Ecology*, 347, 134–143.

Strain E.M.A., Olabarria C., Mayer-Pinto M., Cumbo V., Morris R.L., Bugnot A.B., Dafforn K.A., Heery E., Firth L.B., Brooks P.R. & Bishop M.J. (2018) Eco-engineering urban infrastructure for marine and coastal biodiversity: which interventions have the greatest ecological benefit? *Journal of Applied Ecology*, 55, 426–441.

Williams G.A. & Morritt D. (1995) Habitat partitioning and thermal tolerance in a tropical limpet, *Cellana grata*. *Marine Ecology Progress Series*, 124, 89–103.

2.8. Create 'rock pools' on intertidal artificial structures

- **Eighteen studies** examined the effects of creating 'rock pools' on intertidal artificial structures on the biodiversity of those structures. Ten studies were in estuaries in Australia[1,2,3,4,5,10,11,12], the UK[6a] and eastern USA[9], five were on open coastlines in the UK[6b,6c,7], Ireland[8] and southeast Spain[13], two were in straits in the UK[15] and Malaysia[16], and one was in a marina in Australia[14].

COMMUNITY RESPONSE (17 STUDIES)

- **Overall community composition (16 studies):** Thirteen replicated, controlled studies (including one randomized, six paired sites and three site comparison studies) in Australia[1,2,10,12,14] the UK[6a,6b,6c,7,15], the USA[9], Spain[13] and Malaysia[16], reported that rock pools created on intertidal artificial structures, along with holes in two studies[1,6a], supported macroalgae[1,2,6c,7,9,13,14,15,16], mobile invertebrate[2,6c,7,9,12,13,14,15,16], non-mobile invertebrate[1,2,6a,7,9,14,16] and/or fish[7,9,10,12,15] species that were absent from structure surfaces without pools or holes. One of the studies[13] also found that pools supported different combined macroalgae and invertebrate community composition to surfaces without pools. One replicated, paired sites, controlled study in Australia[5] found mixed effects on the community composition depending on the pool depth, shore level and site. One of the studies[7] found that created pools supported different combined macroalgae and non-mobile invertebrate communities but similar combined mobile invertebrate and fish communities to natural rock pools, while one[12] found that combined mobile invertebrate and fish communities differed to natural pools. Two of the studies[7,16] found that the pool depth did not affect the community composition, while one[14] found that the pool angle did. One replicated study in Ireland[8] found that the shore level and wave-exposure affected the community composition, and that wave-sheltered pools filled with sediment within two years. One replicated, randomized study in Australia[11] found that adding short flexible habitats into pools had mixed effects on community composition depending on the species group and site.

- **Overall richness/diversity (15 studies):** Nine of 12 replicated, controlled studies (including one randomized, six paired sites and two site comparison studies) in Australia[1,2,5,12,14], the UK[6a,6b,6c,7,15], Spain[13] and Malaysia[16] found that rock pools created on intertidal artificial structures, along with holes in two studies[1,6a], supported higher combined macroalgae, invertebrate and/or fish species diversity[13] and/or richness[1,2,6a,6c,7,13,14,15,16] than structure surfaces without pools or holes. Three studies reported similar combined macroalgae and invertebrate[5,6b] or combined mobile invertebrate and fish[12] species richness in pools and on structure surfaces. One of the studies[7] found that combined macroalgae, invertebrate and fish species richness in created pools was similar to natural rock pools, while one[12] reported lower combined mobile invertebrate and fish species richness in created pools. Two of the studies[1,13], along with one replicated study in Ireland[8], found that the shore level of pools, along with holes in one[1], did not affect the species richness, but in one[8], the functional richness (species grouped according to their role in the community) was lower in highshore pools than midshore. Three of the studies[5,7,16] found that the pool depth had no effect on species richness, one[14] found higher richness in tilted pools than horizontal ones, and one replicated, randomized study in Australia[11] found that adding short flexible habitats into pools had mixed effects depending on the species group and site. One before-and-after study in Australia[4] reported that creating pools, along with reducing the slope of a structure, increased the combined macroalgae, invertebrate and fish species richness on the structure.
- **Fish richness/diversity (1 study):** One replicated, paired sites, controlled and site comparison study in Australia[10] reported that creating rock pools on an intertidal artificial structure did not increase the fish species richness on and around the structure.

POPULATION RESPONSE (4 STUDIES)

- **Overall abundance (1 study):** One replicated, randomized study in Australia[11] found that adding short flexible habitats into rock pools created on intertidal artificial structures had mixed effects on macroalgae, invertebrate and fish abundance in pools, depending on the species group and site.
- **Algal abundance (1 study):** One replicated, paired sites, controlled study in Australia[5] found that creating rock pools on intertidal artificial structures had mixed effects on macroalgal abundances depending on the pool depth, shore level, species group and site.
- **Invertebrate abundance (2 studies):** Two replicated, controlled studies (including one with paired sites) in Australia[1,5] found that creating rock pools on intertidal artificial structures, along with holes in one[1], had mixed effects on limpet[1] or combined invertebrate[5] abundances, depending on the shore level[1,5], pool depth[5], species group[5] and/or site[5].
- **Fish abundance (1 study):** One replicated, paired sites, controlled and site comparison study in Australia[10] found that creating rock pools on an intertidal artificial structure had mixed effects on the fish abundance on and around the structure, depending on the species group and site.

BEHAVIOUR (3 STUDIES)

- **Use (2 studies):** Two studies (including one before-and-after study) in Australia[3,4] reported that rock pools created on intertidal artificial structures, along with holes in one study[3], were used by sea slugs[3], urchins[3], octopuses[3], macroalgae[4], invertebrates[4] and fishes[4].
- **Fish behaviour change (1 study):** One replicated, randomized study in Australia[11] found that adding short flexible habitats into rock pools created on intertidal artificial structures did not increase the number of bites fishes took of pool surfaces.

Background

Rock pools provide organisms refuge from desiccation and temperature fluctuations during low tide in intertidal rocky habitats (Williams & Morrit 1995). They are important nursery habitats (Seabra *et al.* 2020) and can extend the vertical range of species along intertidal gradients (Metaxas & Scheibling 1993). They can also, however, become stressful environments under extreme conditions, such as extreme high/low temperatures or salinities (Firth & Williams 2009; Waltham & Sheaves 2020). The size, shape, density and shore level of rock pools is likely to affect the size, abundance and variety of organisms that can use them (Bugnot *et al.* 2018). Large, deep rock pools provide more stable environments than small pools and fishes tend to be more abundant in them. Macroalgae and invertebrates, however, show variable species-specific responses to pool size and depth.

Rock pools often support species that are absent from emergent rock surfaces in both natural and artificial habitats (Firth *et al.* 2013), but are relatively uncommon on artificial structures (Aguilera *et al.* 2014). Water-retaining features mimicking rock pools can be created on intertidal artificial structures by adding or removing material, either during construction or retrospectively.

Definition: 'Rock pools' are depressions with a length to width ratio ≤3:1 and depth ≥50 mm that retain water during low tide (modified from "Intertidal water retaining features" in Strain *et al.* 2018).

See also: *Create natural rocky reef topography on intertidal artificial structures*; *Create hole habitats (>50 mm) on intertidal artificial structures*.

Aguilera M.A., Broitman B.R. & Thiel M. (2014) Spatial variability in community composition on a granite breakwater versus natural rocky shores: lack of microhabitats suppresses intertidal biodiversity. *Marine Pollution Bulletin*, 87, 257–268.

Bugnot A.B., Mayer-Pinto M., Johnston E.L., Schaefer N. & Dafforn K.A. (2018) Learning from nature to enhance blue engineering of marine infrastructure. *Ecological Engineering*, 120, 611–621.

Firth L.B., Thompson R.C., White F.J., Schofield M., Skov M.W., Hoggart S.P.G., Jackson J., Knights A.M. & Hawkins S.J. (2013) The importance of water-retaining features for biodiversity on artificial intertidal coastal defence structures. *Diversity and Distributions*, 19, 1275–1283.

Firth L.B. & Williams G.A. (2009) The influence of multiple environmental stressors on the limpet *Cellana toreuma* during the summer monsoon season in Hong Kong. *Journal of Experimental Marine Biology and Ecology*, 375, 70–75.

Metaxas A. & Scheibling R.E. (1993) Community structure and organization of tidepools. *Marine Ecology Progress Series*, 98, 187–198.

Seabra M.I., Hawkins S.J., Espírito-Santo C., Castro J.J. & Cruz T. (2020) Rock-pools as nurseries for co-existing limpets: spatial and temporal patterns of limpet recruitment. *Regional Studies in Marine Science*, 37, 101339.

Strain E.M.A., Olabarria C., Mayer-Pinto M., Cumbo V., Morris R.L., Bugnot A.B., Dafforn K.A., Heery E., Firth L.B., Brooks P.R. & Bishop M.J. (2018) Eco-engineering urban infrastructure for marine and coastal biodiversity: which interventions have the greatest ecological benefit? *Journal of Applied Ecology*, 55, 426–441.

Waltham N.J. & Sheaves M. (2020) Thermal exposure risks to mobile tropical marine snails: are eco-engineered rock pools on seawalls scale-specific enough for comprehensive biodiversity outcomes? *Marine Pollution Bulletin*, 156, 111237.

Williams G.A. & Morritt D. (1995) Habitat partitioning and thermal tolerance in a tropical limpet, *Cellana grata*. *Marine Ecology Progress Series*, 124, 89–103.

A replicated, controlled study in 2006–2007 on an intertidal seawall in Sydney Harbour estuary, Australia (1) reported that rock pools created on the seawall, along with holes, supported higher macroalgae and non-mobile invertebrate species richness than seawall surfaces without pools or holes, and found that limpet *Siphonaria denticulata* abundance was higher in pools than on surfaces at highshore, but lower at mid and lowshore. After 12–14 months, pools supported 15–37 macroalgae and non-mobile invertebrate species groups/site (highshore: 15–21/site; midshore: 24–36/site; lowshore: 30–37/site), while seawall surfaces without pools or holes supported 2–21/site (highshore: 2–3/site; midshore: 7–11/site; lowshore: 19–21/site) (data not statistically tested). At least 22 species (≥12 macroalgae, ≥10 non-mobile invertebrates) recorded in pools were absent from surfaces without. At highshore, limpets were more abundant in pools (3–59 limpets/pool) than on seawall surfaces (0–8/surface), but the opposite was true at midshore (1–2/pool vs 78–214/surface) and lowshore (0/pool vs 4–10/surface). It is not clear whether these effects were the direct result of creating rock pools or holes. Rock pools were created during July–September 2006 by replacing seawall blocks with water-retaining troughs during construction of a vertical sandstone seawall. Six rectangular pools (length: 600 mm; width: 300 mm; depth: 50 mm; volume: 9 l) were created at highshore, midshore and lowshore in each of three sites along the seawall. Pools were shaded. Pools were compared with six seawall surfaces (length: 600 mm; width: 300 mm) at each shore level and site. Macroalgae and invertebrates were counted in pools and on seawall surfaces during low tide in September 2007.

A replicated, paired sites, controlled study in 2009–2010 on two intertidal seawalls in Sydney Harbour estuary, Australia (2, same experimental set-up as 5) reported that rock pools created on the seawalls supported more macroalgae and invertebrate species than seawall surfaces without rock pools, but data were not statistically tested. After seven months, a total of 33 macroalgae and invertebrate species were recorded in pools compared with 20 on seawall surfaces without pools. Twenty-five species (5 macroalgae, 7 non-mobile invertebrates, 13 mobile invertebrates) recorded in pools were absent from seawall surfaces without. Rock pools were created by attaching concrete pots to vertical sandstone seawalls in December 2009. Six half-flowerpot shaped pools (top diameter: 360 mm; depth: 380 mm; volume: 10 l) were attached at midshore on each of two seawalls. Pools were compared with seawall surfaces (500 × 500 mm) adjacent to each pool. Macroalgae and invertebrates were counted in pools and on seawall surfaces during low tide after seven months. Two pools were missing and no longer provided habitat.

A study (year not reported) on an intertidal seawall in Sydney Harbour estuary, Australia (3) reported that rock pools created on the seawall, along with holes, were used by mobile invertebrates from at least three species groups. Sea slugs (Opistobranchia), urchins (Echinoidea) and octopuses (Octopoda) were recorded in pools and holes. It is not clear whether these effects were the direct result of creating rock pools or holes. Rock pools were created, along with holes, by replacing seawall blocks with sandbags during maintenance of a vertical sandstone seawall, then removing the sandbags to leave shaded water-retaining depressions in the wall. No other details were reported.

A before-and-after study in 2012–2013 on an intertidal seawall in Sydney Harbour estuary, Australia (4) reported that creating rock pools on the seawall, along with reducing the slope of the wall, increased the macroalgae, invertebrate and fish species richness on the wall. A total of 25 macroalgae, invertebrate and fish species were recorded in pools and on the seawall after pools were created and the slope was reduced, compared with 10 species on the seawall before (data not statistically tested). It is not clear whether these effects were the direct result of creating rock pools or reducing the slope of the seawall. However, several macroalgae, invertebrate and fish species recorded in pools were absent from the seawall before pools were created. Rectangular rock pools (area: 2 m^2; depth: 300 mm; volume: 600 l) were created using large rectangular sandstone blocks during reconstruction of a sandstone boulder seawall in July 2012. Pools were lined with pond-liners with limestone gravel and blocks in the base. There were two pools at midshore and one at highshore along the seawall. The slope of the seawall was also reduced during reconstruction. Macroalgae, invertebrates and fishes were counted during low tide on the wall before reconstruction and on the wall and in pools after reconstruction in 2013 (sampling details and month not reported).

A replicated, paired sites, controlled study in 2009–2010 on two intertidal seawalls in Sydney Harbour estuary, Australia (5; same experimental setup as 2) reported that rock pools created on the seawalls supported similar macroalgae and invertebrate species richness to seawall surfaces without pools, but found that community composition and abundances varied depending on the pool depth, shore level, species group and site. After seven months, macroalgae and invertebrate species richness was similar in deep pools (highshore: 7–10 species groups/site; midshore: 16–23/site), shallow pools (highshore: 11/site; midshore: 23/site) and on seawall surfaces without pools (highshore: 10/site; midshore: 12–18/site) (data not statistically tested). Abundances and community composition varied in pools and on seawall surfaces depending on the pool depth, shore level and site (see paper for results). Rock pools were created by attaching concrete pots to vertical sandstone seawalls in December 2009. Six deep (depth: 380 mm; volume: 10 l) and six shallow (220 mm; 6 l) half-flowerpot shaped pools (top diameter: 360 mm) were attached at both highshore and midshore on each of two seawalls. Pools were compared with seawall surfaces adjacent to each pool with surface areas matching inside pool surfaces (deep: 500 × 500 mm, reported from 2; shallow: not reported). Macroalgae and invertebrates were counted in pools and on seawall surfaces during low tide over seven months. Five deep and 10 shallow pools were missing and no longer provided habitat.

A replicated, controlled study in 2010–2011 on an intertidal seawall in the Teign estuary, UK (6a) found that rock pools created on the seawall, along with holes, supported higher macroalgae and invertebrate species richness than seawall surfaces without pools or holes. After 19 months, macroalgae and invertebrate species richness was higher in pools (3 species/pool) than on seawall surfaces without (1/surface). Barnacles (Cirripedia) were recorded only in pools. It is not clear whether these effects were the direct result of creating rock pools or holes. Rock pools were created in May 2010 by replacing seawall blocks with water-retaining troughs during construction of a vertical sandstone seawall. Fifteen square pools (150 × 150 mm; depth/volume not reported) were created at highshore. Pools were shaded. Pools were compared with 15 mortar seawall surfaces (150 × 150 mm). Macroalgae and invertebrates were counted in pools and on seawall surfaces during low tide after 19 months. Three pools and seven surfaces had been buried by sediment and no longer provided habitat.

A replicated, controlled study in 2012–2013 on an intertidal groyne on open coastline in the Irish Sea, UK (6b) reported that rock pools created on a concrete block placed in the groyne supported similar macroalgae and invertebrate species richness to groyne surfaces without pools. Data were not statistically tested. After 13 months, a total of five species were recorded in large deep pools, four in small deep pools, three in each of large and small shallow pools, and four on groyne surfaces without pools. Rock pools were created in the top surface of a concrete block (1.5 × 1.5 × 1 m) using a mould. Cylindrical pools were either large (diameter: 250 mm) or small (150 mm), and either deep (depth: 200 mm) or shallow (100 mm). There were three of each size-depth combination. The block was placed at midshore in a boulder groyne during construction in February 2012. Pools were compared with horizontal surfaces of adjacent groyne boulders (dimensions/material not reported). Macroalgae and invertebrates were counted in pools and on groyne surfaces during low tide over 13 months.

A replicated, paired sites, controlled study in 2012–2013 on an intertidal breakwater on open coastline in the Irish Sea, UK (6c) found that rock pools created on the breakwater supported higher macroalgae and invertebrate species richness than breakwater surfaces without pools. After nine months, macroalgae and invertebrate species richness was higher in pools (4 species/pool) than on breakwater surfaces without (2/surface). Two species groups (1 macroalgae, 1 mobile invertebrate) recorded in pools were absent from breakwater surfaces. Rock pools were created on a boulder breakwater in June 2012 by pouring concrete into existing core holes in breakwater boulders. Nine cylindrical pools were created (depth: 100 mm; diameter/shore level not reported). Pools were compared with horizontal surfaces on breakwater boulders adjacent to each pool with surface areas matching inside pool surfaces (dimensions/material not reported). Macroalgae and invertebrates were counted in pools and on breakwater surfaces during low tide after nine months. Four pools leaked water and did not provide rock pool habitat.

A replicated, paired sites, controlled and site comparison study in 2012–2013 on an intertidal breakwater on open coastline in the Irish Sea, UK (7) found that rock pools created on the breakwater supported higher macroalgae, invertebrate and fish species richness than breakwater surfaces without pools, and similar species richness but different community composition to natural rock pools. After 18 months, a total of 23 macroalgae, invertebrate and fish species were recorded in created pools and 14 on breakwater surfaces without pools (data not statistically tested). Community composition (data reported as statistical model results) and average species richness was similar in deep and shallow created pools (both 8 species/pool), and richness was higher in both than on surfaces without (6/surface). Twenty species (7 macroalgae, 6 mobile invertebrates, 6 non-mobile invertebrates, 1 fish) recorded in pools over 18 months were absent from breakwater surfaces. Species richness and the mobile invertebrate and fish community composition were similar in created and natural pools, but the macroalgae and non-mobile invertebrate community composition differed (data reported as statistical model results). Rock pools were created in April 2012 by drilling into horizontal surfaces of a granite boulder breakwater using a core-drill. Nine deep (depth: 120 mm; volume 2.1 l) and nine shallow (50 mm; 0.9 l) cylindrical pools (diameter: 150 mm) were drilled at midshore. Pools were compared with breakwater surfaces on horizontal and vertical boulder surfaces adjacent to each pool, with surface areas matching the inside pool surfaces (deep-vertical: 230 × 230 mm; shallow-vertical: 150 × 150 mm; both-horizontal: 130 × 130 mm; A. Evans *pers. comms.*), and also with natural rock pools on three nearby reefs. Both were pre-cleared of organisms. Macroalgae, invertebrates and fishes were counted in pools and on breakwater surfaces during low tide over 18 months.

A replicated study in 2013–2015 on an intertidal causeway on open coastline in Galway Bay, Ireland (8) reported that rock pools created on the wave-exposed side of the causeway were used by macroalgae, invertebrates and fish, but that pools created on the wave-sheltered side filled with sediment and failed to provide rock pool habitat. After 24 months, a total of 72 macroalgae, invertebrate and fish species groups (highshore: 37; midshore: 63) from 11 functional groups (highshore: 10; midshore: 11) were recorded in pools on the wave-exposed side of the causeway (data not statistically tested). Average species richness was similar in highshore (14 species groups/pool) and midshore (17/pool) pools, but the community composition differed (data reported as statistical model results). The average number of functional groups was lower in highshore (7/pool) than midshore (9/pool) pools. Rock pools were created by pouring concrete around buckets in the base of Shepherd Hill Energy Dissipation units on a causeway, then removing the buckets to leave bucket-shaped pools (top diameter: 130–140 mm; bottom diameter: 110 mm; depth 100–120 mm). Twenty pools were created at both highshore and midshore on each side of the causeway (wave-exposed, wave-sheltered) in June 2013. Macroalgae, invertebrates and fishes in pools were counted during low tide over 24 months and in the laboratory after 24 months. Species were grouped into functional groups according to their role in the community (shape/structure and feeding strategy). All pools on the sheltered side had filled with sediment and no longer provided rock pool habitat.

A replicated, controlled study in 2013–2014 on an intertidal seawall in the Hudson River estuary, USA (9) reported that rock pools created on the seawall supported macroalgae, invertebrate and fish species that were absent from seawall surfaces without rock pools. After nine months, pools supported 89–100% cover of macroalgae and non-mobile invertebrates and at least seven species (1 macroalgae, 2 non-mobile invertebrates, 3 mobile invertebrates, ≥1 fish) that were absent from seawall surfaces without pools. Rock pools were created by placing concrete troughs amongst a boulder seawall during construction. Seven rectangular pools (volume: 59 l; other dimensions not reported) with stepped sides were installed at highshore in November 2013. Macroalgae, invertebrates and fishes were counted during low tide in pools and on surrounding seawall surfaces at the same shore level (details not reported) after nine months.

A replicated, paired sites, controlled and site comparison study in 2014–2015 on three intertidal seawalls in Sydney Harbour estuary, Australia (10) found that creating rock pools on one seawall did not increase the fish species richness on and around the wall, but had mixed effects on fish abundances depending on the species group and site. Over the first 12 months, pelagic fish species richness was similar around the seawall with rock pools (1–3 species/survey) and those without (1–2/survey) and there were no clear differences in maximum abundances (0–26 vs 0–13 individuals/survey), which varied by species group and site (see paper for results). For the seawall with pools, pelagic fish species richness in and around pools (1–2 species/survey) was similar to seawall surfaces without (2–4/survey), and maximum abundances varied by species group (with pools: 0–39 individuals/survey; without: 0–30/survey). After 15–21 months, benthic fish species richness in and around pools was similar to seawall surfaces without (both 9 species in total). Total abundance was higher for pools than surfaces for one species (12 vs 1 individuals), but similar for 10 others (0–15 vs 0–22). One species recorded in and around pools was absent from surfaces without. Rock pools were created by attaching concrete pots to a vertical sandstone seawall in February 2014. Five half-flowerpot shaped pools (top diameter: 315 mm; volume: 7l; depth not reported) were attached at midshore in each of two sites along the seawall. Pools were compared with seawall surfaces without pools (dimensions not reported) adjacent to each site and on two other seawalls without pools. Fishes were counted in, on and around pools and seawall surfaces from time-lapsed photographs and videos during 7–10 high tides over 21 months.

A replicated, randomized study in 2016 on two intertidal seawalls in Sydney Harbour estuary, Australia (11) reported that rock pools created on the seawalls supported macroalgae, invertebrates and fishes, and found that adding short flexible habitats (coir panels) to pools had mixed effects on community composition, species richness and abundances depending on the species group and site. Over eight months, during low tide, a total of 44 macroalgae, invertebrate and fish species groups were recorded in pools with coir and 57 in pools without (data not statistically tested). Average macroalgae and non-mobile invertebrate species richness was lower in pools with coir (9 species/pool) than without (12/pool) and the community composition differed (data reported as statistical model results), while abundances varied depending on the species group and site (data not reported). Mobile invertebrate and fish species richness was also lower in pools with coir (2 species/pool) than without (3/pool), but their abundance was similar (data not reported), while effects on their community composition varied by site. During high tide, a total of 13 fish species were recorded in and around pools with coir and 14 in and around pools without, while 49 mobile invertebrate species groups were recorded in each. Average fish species richness, abundance, community composition, and the number of bites they took, were all similar in pools with and without coir (data not reported). Mobile invertebrate species richness in pools with coir (8–11 species/pool) and without (9–16/pool) varied by site, but the community composition was similar. Rock pools were created in January–February 2016 by attaching concrete pots to two vertical sandstone seawalls. Five half-flowerpot shaped pools (top diameter: 315 mm; depth: 300 mm; volume: 7 l) with coir panels on one inside surface and five without were randomly arranged at midshore in each of two sites along each seawall. Macroalgae, invertebrates and fishes were counted in pools during low tide over eight months. Mobile invertebrates and fishes were also surveyed during two high tides using a suction pump and videos, respectively. Three pools were missing and no longer provided habitat.

A replicated, controlled and site comparison study in 2013–2015 on two intertidal seawalls in Sydney Harbour estuary, Australia (12) reported that rock pools created on the seawalls supported similar mobile invertebrate and fish species richness to seawall surfaces without pools, and different community composition with lower richness compared with natural rock pools. Over 18 months, a total of 10 mobile invertebrate and fish species groups were recorded in created pools, 10 on seawall surfaces without pools, and 32 in natural pools (data not statistically tested). Five species groups (4 mobile invertebrates, 1 fish) recorded in created pools were absent from seawall surfaces without pools. After 18 months, community composition differed in created and natural pools (data reported as statistical model results). Rock pools were created in December 2013 by attaching concrete pots to two vertical concrete and sandstone seawalls. Five half-flowerpot shaped pools (top diameter: 315 mm; volume: 7 l; depth not reported) were attached at midshore on each seawall. Pools were compared with seawall surfaces (details not reported) and with natural rock pools, cleared of organisms, on a nearby reef. Mobile invertebrates and fishes were counted in three created and natural pools and on seawall surfaces during low tide over 18 months.

A replicated, paired sites, controlled study in 2014–2015 on one intertidal breakwater and two groynes on open coastline in the Alboran Sea, Spain (13) found that rock pools created on the structures supported different macroalgae and invertebrate community composition with higher species diversity and richness than structure surfaces without pools. After 12 months, macroalgae and invertebrate species diversity (data reported as Shannon index) and richness were higher in pools (9 species/pool) than on structure surfaces without (6/surface), and the community composition differed (data reported as statistical model results). Upper-midshore pools supported similar richness to lower-midshore ones (8 vs 9 species/pool). Eight species (4 macroalgae, 4 mobile invertebrates) recorded in pools were absent from structure surfaces without. Rock pools were created in February 2014 by drilling into horizontal surfaces of three limestone boulder structures (1 breakwater, 2 groynes; treated as 1 site) using a jackhammer. Five irregularly-shaped pools (average length: 176 mm; width: 137 mm; depth: ≤20 mm; volume: 0.4 l) were drilled at both upper-midshore and lower-midshore on each structure. Pools were compared with breakwater/groyne surfaces (200 × 200 mm) adjacent to each pool. Macroalgae and invertebrates were counted in pools and on structure surfaces from photographs over 12 months.

A replicated, randomized, controlled study in 2014–2016 on an intertidal seawall in a marina in the Coral Sea, Australia (14) reported that rock pools created on the seawall supported higher macroalgae and invertebrate species richness than seawall surfaces without pools, and that tilted pools supported different community composition with higher species richness than horizontal ones. Over 24 months, a total of 16 macroalgae and invertebrate species groups were recorded in both landward- and seaward-tilting pools, 11 in horizontal pools, and 10 on seawall surfaces without pools (data not statistically tested). Community composition was similar in landward- and seaward-tilted pools, but both differed to horizontal pools (data reported as statistical model results). Ten species (2 macroalgae, 5 mobile invertebrates, 3 non-mobile invertebrates) recorded in pools were absent from seawall surfaces. Sediment accumulation was similar in all pools (34–45 mm depth). Rock pools were created by attaching concrete troughs to a boulder seawall in June 2014. Rectangular pools (length: 400 mm; width: 250 mm; depth: 350 mm; volume: 35 l) were either horizontal or tilted 45° towards the land or sea, thus shaded. There were three of each randomly arranged at midshore along the seawall. Macroalgae and invertebrates were counted in pools and on seawall surfaces (details not reported) during low tide over 24 months. One horizontal pool was missing and no longer provided habitat.

A replicated, controlled study in 2013–2018 on an intertidal seawall in the Solent strait, UK (15) found that rock pools created on the seawall supported higher macroalgae, invertebrate and fish species richness than seawall surfaces without pools. After five years, macroalgae, invertebrate and fish species richness was higher in and on pools (10 species/pool) than on seawall surfaces without (7/surface). Fourteen species (6 macroalgae, 6 mobile invertebrates, 1 fish) recorded in pools over five years were absent from seawall surfaces without. Rock pools were created in September 2013 by attaching concrete pots (Vertipools™) to a vertical concrete seawall. Five triangular Vertipools™ (length: 800 mm; width: 300 mm; depth: 10–200 mm; volume: 10 l) were attached at mid-highshore. Pools were compared with five seawall surfaces with surface areas matching the inside and outside pool surfaces (500 × 500 mm). Macroalgae, invertebrates and fishes were counted during low tide in and on pools (averaged over inside and outside surfaces) on 10 occasions and on seawall surfaces on four occasions over five years.

A replicated, paired sites, controlled study in 2015–2018 on three intertidal seawalls in Penang Strait, Malaysia (16) found that rock pools created on the seawalls supported higher macroalgae and invertebrate species richness than seawall surfaces without pools. After 36 months, a total of 14 macroalgae and invertebrate species were recorded in pools and six on seawall surfaces without (data not statistically tested). Average species richness was higher in pools (13 species/pool) than on seawall surfaces (6/surface). Community composition (data reported as statistical model results) and species richness were similar in deep and shallow pools (both 11 species/pool). Thirteen species (1 macroalgae, 11 mobile invertebrates, 1 non-mobile invertebrate) recorded in pools over 36 months were absent from seawall surfaces. Rock pools were created in October 2015 by drilling into horizontal surfaces of three granite boulder seawalls using a core-drill. Fifteen deep (depth: 120 mm; volume: 2.1 l) and 15 shallow (50 mm; 0.9 l) cylindrical pools (diameter: 150 mm) were drilled at midshore on each seawall. Pools were compared with seawall surfaces, cleared of organisms, adjacent to each pool, with surface areas matching the inside pool surfaces (deep: 270 × 270 mm; shallow: 200 × 200 mm). Macroalgae and invertebrates were counted in pools and on seawall surfaces during low tide over 36 months.

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2.9. Create small adjoining cavities or 'swimthrough' habitats (≤100 mm) on intertidal artificial structures

- **Two studies** examined the effects of creating small adjoining cavities or 'swimthrough' habitats on intertidal artificial structures on the biodiversity of those structures. One study was on an open coastline in the UK and in an estuary in the Netherlands[1] and one was on an open coastline in South Africa[2].

COMMUNITY RESPONSE (2 STUDIES)

- **Invertebrate community composition (1 study):** One replicated, controlled study in South Africa[2] found that creating small swimthrough habitats on intertidal artificial structures did not alter the mobile invertebrate community composition on structure surfaces.
- **Overall richness/diversity (1 study):** One replicated study in the UK and the Netherlands[1] found that varying the size and arrangement of small swimthrough habitats created on intertidal artificial structures did not increase the combined macroalgae and invertebrate species richness in and on the structures.
- **Invertebrate richness/diversity (1 study):** One replicated, controlled study in South Africa[2] found that creating small swimthrough habitats on intertidal artificial structures did not increase the mobile invertebrate species richness or diversity on structure surfaces.

POPULATION RESPONSE (2 STUDIES)

- **Invertebrate abundance (2 studies):** One replicated, controlled study in South Africa[2] found that creating small swimthrough habitats on intertidal artificial structures increased the mobile invertebrate abundance on structure surfaces. One replicated study in the UK and the Netherlands[1] found that varying the size and arrangement of small swimthrough habitats altered the invertebrate abundance in and on structures.

BEHAVIOUR (0 STUDIES)

Background

Small adjoining cavities or 'swimthrough' habitats are not well-studied in intertidal rocky habitats. They may form through weathering of softer rocks, amongst loosely-consolidated cobbles/boulders, or amongst three-dimensional structures created by living organisms. They likely provide organisms refuge from desiccation, temperature fluctuations and predation, in the same way crevice, hole and rock pool habitats do (Menge & Lubchenco 1981; Williams & Morritt 1995). They could also serve as corridors, connecting adjacent refuge habitats. The size and density of cavities or swimthroughs is likely to affect the size, abundance and variety of organisms that can use them. Small habitats can provide refuge for small-bodied organisms but may exclude larger organisms, limit their growth and get rapidly filled-up (Firth *et al.* 2020). Large habitats can be used by larger-bodied organisms but may not provide sufficient refuge from predators for smaller organisms. By default, cavities and swimthroughs contain shaded surfaces, which can be associated with the presence of non-native species (Dafforn 2017).

Cavities/swimthroughs are sometimes present on marine artificial structures made of consolidated boulders or blocks (Sherrard *et al.* 2016) or gabion baskets (Firth *et al.* 2014), but are absent from many other structures. Small adjoining cavities or 'swimthrough' habitats can be created on intertidal artificial structures by adding or removing material, either during construction or retrospectively.

Definition: 'Small adjoining cavities or 'swimthrough' habitats' are adjoining internal cavities sheltered from, but with access to/from, outside the structure. Dimensions can vary but are ≤100 mm in any direction.

See also: *Create hole habitats (>50 mm) on intertidal artificial structures*; *Create crevice habitats (>50 mm) on intertidal artificial structures*; *Create 'rock pools' on intertidal artificial structures*; *Create large adjoining cavities or 'swimthrough' habitats (>100 mm) on intertidal artificial structures*.

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Williams G.A. & Morritt D. (1995) Habitat partitioning and thermal tolerance in a tropical limpet, *Cellana grata*. *Marine Ecology Progress Series*, 124, 89–103.

A replicated study in 2011–2014 on 30 rock gabions on open coastline in the Irish Sea, UK and in the Eastern Scheldt estuary, Netherlands (1) found that gabions with small swimthrough habitats created amongst rocks of mixed sizes (small and large) supported similar macroalgae and invertebrate species richness and abundance to those amongst rocks of regular sizes (small or large), but that abundance was higher on gabions with regular small rocks than regular large ones. In the UK, after 12 months, 12 mobile and non-mobile invertebrate species were recorded in and on gabions with swimthroughs. Species richness and abundance was similar in and on gabions with swimthroughs amongst mixed-sized rocks (9 species/gabion, 252 individuals/gabion) and regularly-sized ones (8 species/gabion, 191–366 individuals/gabion). Abundance was higher in and on gabions with small regular rocks (366 individuals/gabion) than large regular ones (191/gabion). In the Netherlands, after 16 months, 14 macroalgae, mobile and non-mobile invertebrate species were recorded and overall species richness was similar on all gabion designs (data not reported). Small swimthrough habitats were created amongst rocks in gabion baskets (500 × 500 × 300 mm; 76 mm mesh size). Swimthroughs were either amongst rocks of mixed sizes (small: 60–100 mm and large: 180 mm), or amongst regularly-sized small or large rocks. Five of each design were placed at midshore on a boulder beach in the UK in April 2011 and at lower-midshore on a sandy beach in the Netherlands in September 2012. UK gabions were dismantled and invertebrates counted after 12 months. In the Netherlands, macroalgae and invertebrates on external horizontal gabion surfaces were counted after 16 months.

A replicated, controlled study (year not reported) in two intertidal boulder fields on open coastline in the Indian Ocean, South Africa (2) found that small swimthrough habitats created under concrete blocks supported similar mobile invertebrate species richness, diversity and community composition to blocks without swimthroughs, but higher mobile invertebrate abundance. Swimthrough habitats supported similar mobile invertebrate species richness, diversity and community composition (data reported as statistical model results) but higher mobile invertebrate abundance (3 individuals/dm^2) compared with blocks without swimthroughs (1/dm^2). Small swimthrough habitats (length: 290 mm; width: 70 mm; height: 20 mm) were created on the undersides of concrete blocks (250 × 150 × 40 mm) using a mould. Twelve blocks with swimthroughs and 12 without were placed on the seabed in each of two natural boulder-fields (material, shore level and month/year not reported). Mobile invertebrates on the horizontal surface (roof) of swimthrough habitats and on the equivalent undersurface of blocks without swimthroughs were counted from photographs after seven weeks. Some blocks were missing and no longer provided habitat (numbers not reported).

(1) Firth L.B., Thompson R.C., Bohn K., Abbiati M., Airoldi L., Bouma T.J., Bozzeda F., Ceccherelli V.U., Colangelo M.A., Evans A., Ferrario F., Hanley M.E., Hinz H., Hoggart S.P.G., Jackson J.E., Moore P., Morgan E.H., Perkol-Finkel S., Skov M.W., Strain E.M., van Belzen J. & Hawkins S.J. (2014) Between a rock and a hard place: environmental and engineering considerations when designing coastal defence structures. *Coastal Engineering*, 87, 122–135.

(2) Liversage K., Cole V., Coleman R. & McQuaid C. (2017) Availability of microhabitats explains a widespread pattern and informs theory on ecological engineering of boulder reefs. *Journal of Experimental Marine Biology and Ecology*, 489, 36–42.

2.10. Create large adjoining cavities or ‘swimthrough’ habitats (>100 mm) on intertidal artificial structures

- We found no studies that evaluated the effects of creating large adjoining cavities or ‘swimthrough’ habitats on intertidal artificial structures on the biodiversity of those structures.

This means we did not find any studies that directly evaluated this intervention during our literature searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Background

Large adjoining cavities or ‘swimthrough’ habitats are not well-studied in intertidal rocky habitats. They may form through weathering of softer rocks, amongst loosely-consolidated boulders, or within three-dimensional structures created by living organisms. They likely provide organisms refuge from desiccation, temperature fluctuations and predation, in the same way crevice, hole and rock pool habitats do (Menge & Lubchenco 1981; Williams & Morritt 1995). They could also serve as corridors, connecting adjacent refuge habitats. The size and density of cavities or swimthroughs is likely to affect the size, abundance and variety of organisms that can use them. Small habitats can provide refuge for small-bodied organisms but may exclude larger organisms, limit their growth and get rapidly filled-up (Firth *et al.* 2020). Large habitats can be used by larger-bodied organisms but may not provide sufficient refuge from predators for smaller organisms. By default, cavities and swimthroughs contain shaded surfaces, which can be associated with the presence of non-native species (Dafforn 2017).

Cavities/swimthroughs are sometimes present on marine artificial structures made of consolidated boulders or blocks (Sherrard *et al.* 2016) or gabion baskets (Firth *et al.* 2014), but are absent from many other structures. Large adjoining cavities or ‘swimthrough’ habitats can be created on intertidal artificial structures by adding or removing material, either during construction or retrospectively.

Definition: ‘Large adjoining cavities or ‘swimthrough’ habitats’ are adjoining internal cavities sheltered from, but with access to/from, outside the structure. Dimensions can vary but are >100 mm in any direction.

See also: *Create hole habitats (>50 mm) on intertidal artificial structures*; *Create crevice habitats (>50 mm) on intertidal artificial structures*; *Create ‘rock pools’ on intertidal artificial structures*; *Create small adjoining cavities or ‘swimthrough’ habitats (≤100 mm) on intertidal artificial structures*.

Dafforn K.A. (2017) Eco-engineering and management strategies for marine infrastructures to reduce establishment and dispersal of non-indigenous species. *Management of Biological Invasions*, 8, 153–161.

Firth L.B., Airoldi L., Bulleri F., Challinor S., Chee S.-Y., Evans A.J., Hanley M.E., Knights A.M., O’Shaughnessy K., Thompson R.C. & Hawkins S.J. (2020) Greening of grey infrastructure should not be used as a Trojan horse to facilitate coastal development. *Journal of Applied Ecology*, 57, 1762–1768.

Firth L.B., Thompson R.C., Bohn K., Abbiati M., Airoldi L., Bouma T.J., Bozzeda F., Ceccherelli V.U., Colangelo M.A., Evans A., Ferrario F., Hanley M.E., Hinz H., Hoggart S.P.G., Jackson J.E., Moore P., Morgan E.H.,

Perkol-Finkel S., Skov M.W., Strain E.M., van Belzen J. & Hawkins S.J. (2014) Between a rock and a hard place: environmental and engineering considerations when designing coastal defence structures. *Coastal Engineering*, 87, 122–135.
Menge B.A. & Lubchenco J. (1981) Community organization in temperate and tropical rocky intertidal habitats: prey refuges in relation to consumer pressure gradients. *Ecological Monographs*, 51, 429–450.
Sherrard T.R.W., Hawkins S.J., Barfield P., Kitou M., Bray S. & Osborne P.E. (2016) Hidden biodiversity in cryptic habitats provided by porous coastal defence structures. *Coastal Engineering*, 118, 12–20.
Williams G.A. & Morritt D. (1995) Habitat partitioning and thermal tolerance in a tropical limpet, *Cellana grata*. *Marine Ecology Progress Series*, 124, 89–103.

2.11. Create small protrusions (1–50 mm) on intertidal artificial structures

- **Two studies** examined the effects of creating small protrusions on intertidal artificial structures on the biodiversity of those structures. Both studies were on island coastlines in the Singapore Strait[1a,1b].

COMMUNITY RESPONSE (2 STUDIES)

- **Overall community composition (2 studies):** One of two replicated, randomized, controlled studies in Singapore[1a,1b] found that creating small protrusions on intertidal artificial structures did not alter the combined macroalgae and invertebrate community composition on structure surfaces[1a]. One study[1b] found that creating small protrusions, along with grooves, small ridges and pits, had mixed effects on the community composition, depending on the site and the size and arrangement of protrusions and other habitats.
- **Overall richness/diversity (2 studies):** Two replicated, randomized, controlled studies in Singapore[1a,1b] found that creating small protrusions on intertidal artificial structures, along with grooves, small ridges and pits in one study[1b], increased the combined macroalgae and invertebrate species richness on structure surfaces. One of the studies[1b] found that varying the size and arrangement of protrusions and other habitats had mixed effects on species richness, depending on the shore level.

POPULATION RESPONSE (2 STUDIES)

- **Overall abundance (2 studies):** One of two replicated, randomized, controlled studies in Singapore[1a,1b] found that creating small protrusions on intertidal artificial structures did not increase the combined macroalgae and invertebrate abundance on structure surfaces[1a]. One study[1b] found that creating small protrusions, along with grooves, small ridges and pits, had mixed effects on abundance, depending on the shore level, site, and the size and arrangement of protrusions and other habitats.

BEHAVIOUR (0 STUDIES)

Background

Small protrusions create vertical or horizontal (i.e. overhangs) relief in intertidal rocky habitats. They can provide organisms refuge from desiccation and temperature fluctuations during low tide (Williams & Morrit 1995) and also shelter from predation or grazing (Wahl & Hoppe 2002). Some species preferentially recruit to habitats with high vertical or horizontal relief (Harmelin-Vivien *et al.* 1995). The size and density of protrusions is likely to affect the size, abundance and variety of organisms that can use them. Small habitats can provide refuge for small-bodied organisms but may exclude larger organisms and limit their growth. Large habitats can be used by larger-bodied organisms but may not provide sufficient refuge from predators for smaller organisms. By default, horizontal protrusions (overhangs) create shaded and downward-facing surfaces, which can be associated with the presence of non-native species (Dafforn 2017).

Protrusions are sometimes present on quarried boulders used in marine artificial structures (MacArthur *et al.* 2020) but are often absent from other types of structures. Small protrusions can be created on intertidal artificial structures by adding material, either during construction or retrospectively.

Definition: 'Small protrusions' are elevations with a length to width ratio ≤3:1 that protrude 1–50 mm from the substratum (modified from "Small elevations" in Strain *et al.* 2018).

See also: *Create textured surfaces (≤1 mm) on intertidal artificial structures*; *Create natural rocky reef topography on intertidal artificial structures*; *Create large protrusions (>50 mm) on intertidal artificial structures*; *Create small ridges or ledges (1–50 mm) on intertidal artificial structures*; *Create large ridges or ledges (>50 mm) on intertidal artificial structures*; *Create groove habitats* and *small protrusions, ridges or ledges (1–50 mm) on intertidal artificial structures*.

Dafforn K.A. (2017) Eco-engineering and management strategies for marine infrastructures to reduce establishment and dispersal of non-indigenous species. *Management of Biological Invasions*, 8, 153–161.

Harmelin-Vivien M.L., Harmelin J.G. & Leboulleux V. (1995) Microhabitat requirements for settlement of juvenile sparid fishes on Mediterranean rocky shores. *Hydrobiologia*, 300, 309–320.

MacArthur M., Naylor L.A., Hansom J.D. & Burrows M.T. (2020) Ecological enhancement of coastal engineering structures: passive enhancement techniques. *Science of the Total Environment*, 740, 139981.

Strain E.M.A., Olabarria C., Mayer-Pinto M., Cumbo V., Morris R.L., Bugnot A.B., Dafforn K.A., Heery E., Firth L.B., Brooks P.R. & Bishop M.J. (2018) Eco-engineering urban infrastructure for marine and coastal biodiversity: which interventions have the greatest ecological benefit? *Journal of Applied Ecology*, 55, 426–441.

Wahl M. & Hoppe K. (2002) Interactions between substratum rugosity, colonization density and periwinkle grazing efficiency. *Marine Ecology Progress Series*, 225, 239–249.

Williams G.A. & Morritt D. (1995) Habitat partitioning and thermal tolerance in a tropical limpet, *Cellana grata*. *Marine Ecology Progress Series*, 124, 89–103.

A replicated, randomized, controlled study in 2009–2010 on two intertidal seawalls on island coastlines in the Singapore Strait, Singapore (1a; same experimental set-up as 1b) found that concrete settlement plates with small protrusions supported higher macroalgae and invertebrate species richness but similar community composition and abundances compared with granite plates without protrusions. After 13 months, macroalgae and invertebrate species richness was higher on settlement plates with small protrusions (8 species/plate) than without (3/plate), while abundances were statistically similar (88 vs 178 individuals/plate). Community composition was similar on plates with and without protrusions (data reported as statistical model results). Settlement plates (200 × 200 mm) were moulded with and without small protrusions. Plates with protrusions were concrete with 36 cylindrical protrusions/plate with either regular (16 mm width, height and spacing) or variable (4–28 mm) arrangement. Plates without protrusions were granite fragments set in cement. Granite may be considered an environmentally-sensitive material compared with concrete (see "*Use environmentally-sensitive material on intertidal artificial structures*"). Eight of each design were randomly-arranged at both lowshore and highshore on each of two granite boulder seawalls in November–December 2009. Macroalgae on plates were counted from photographs and invertebrates in the laboratory after 13 months.

A replicated, randomized, controlled study in 2009–2010 on two intertidal seawalls on island coastlines in the Singapore Strait, Singapore (1b; same experimental set-up as 1a) found that concrete settlement plates with small protrusions, along with grooves, small ridges and pits, supported higher macroalgae and invertebrate species richness than granite plates without added habitats, but that abundances and community composition varied depending on the habitat arrangement, shore level and site. After 13 months, macroalgae and invertebrate species richness was higher on settlement plates with protrusions, grooves, ridges and pits than without at lowshore (13–23 vs 6–10 species/plate) and highshore (5–9 vs 2–3/plate). Richness was higher on plates with variable habitats than regular ones at lowshore (22–23 vs 13–16/plate), but not highshore (6–9 vs 5–6/plate). Abundances were higher on plates with added habitats than without in four of eight comparisons (9–833 vs 3–208 individuals/plate), while community composition differed in three of four comparisons (data reported as statistical model results). In all other comparisons, results were similar (abundances: 104–1,957 vs 49–1,162/plate). It is not clear whether these effects were the direct result of creating protrusions, grooves, ridges or pits. However, plate quarters with protrusions had similar richness (8 species/quarter) and abundances (88 individuals/quarter) to quarters with the other habitat types (6–11 species and 97–231 individuals/quarter). Settlement plates (400 × 400 mm) were moulded with and without small protrusions, along with grooves, small ridges and pits. Plates with added habitats were concrete. Each 200 × 200 mm quarter contained either 36 cylindrical protrusions, four-to-five grooves and ridges, 12 ridges or 36 pits. All habitats had either regular (16 mm width, depth/height and spacing) or variable (4–28 mm) arrangement. Plates without added habitats were granite fragments set in cement. Granite may be considered an environmentally-sensitive material compared with concrete (see "*Use environmentally-sensitive material on intertidal artificial structures*"). Eight of each design were randomly arranged at both lowshore and highshore on each of two granite boulder seawalls in November–December 2009. Macroalgae on plates were counted from photographs and invertebrates in the laboratory after 13 months.

(1) Loke L.H.L. & Todd P.A. (2016) Structural complexity and component type increase intertidal biodiversity independently of area. *Ecology*, 97, 383–393.

2.12. Create large protrusions (>50 mm) on intertidal artificial structures

- **Two studies** examined the effects of creating large protrusions on intertidal artificial structures on the biodiversity of those structures. One study was on an open coastline in the UK[1] and one was in a marina in northeast Australia[2].

COMMUNITY RESPONSE (2 STUDIES)

- **Overall community composition (1 study):** One replicated, randomized, controlled study in Australia[2] reported that large protrusions created on an intertidal artificial structure supported mobile and non-mobile invertebrate species that were absent from structure surfaces without protrusions. The study also found that protrusions tilted at an angle supported different combined macroalgae and invertebrate community composition to horizontal ones.
- **Overall richness/diversity (2 studies):** Two replicated, controlled studies (including one randomized study) in the UK[1] and Australia[2] found that creating large protrusions on an intertidal artificial structure, along with large ridges in one study[1], did not increase the combined macroalgae and invertebrate species richness on structure surfaces. One of the studies[2] also reported that tilting protrusions at an angle did not increase the species richness compared to those that were horizontal.

POPULATION RESPONSE (1 STUDY)

- **Invertebrate abundance (1 study):** One replicated, controlled study in the UK[1] found that creating large protrusions on an intertidal artificial structure, along with large ridges, increased limpet but not barnacle abundance on structure surfaces.

BEHAVIOUR (0 STUDIES)

Background

Large protrusions create vertical or horizontal (i.e. overhangs) relief in intertidal rocky habitats. They can provide organisms refuge from desiccation and temperature fluctuations during low tide (Williams & Morrit 1995) and alter flow velocities (Guichard & Bourget 1998). Some species preferentially recruit to habitats with high vertical or horizontal relief, potentially to avoid predators (Harmelin-Vivien *et al.* 1995). The size and density of protrusions is likely to affect the size, abundance and variety of organisms that can use them. Small habitats can provide refuge for small-bodied organisms but may exclude larger organisms and limit their growth. Large habitats can be used by larger-bodied organisms but may not provide sufficient refuge from predators for smaller organisms. By default, horizontal protrusions (overhangs) create shaded and downward-facing surfaces, which can be associated with the presence of non-native species (Dafforn 2017).

Protrusions are sometimes present on quarried boulders used in marine artificial structures (MacArthur *et al.* 2020) but are often absent from other types of structures.

Large protrusions can be created on intertidal artificial structures by adding material, either during construction or retrospectively.

Definition: 'Large protrusions' are elevations with a length to width ratio ≤3:1 that protrude >50 mm from the substratum (modified from "Large elevations" in Strain *et al.* 2018).

See also: *Create textured surfaces (≤1 mm) on intertidal artificial structures*; *Create natural rocky reef topography on intertidal artificial structures*; *Create small protrusions (1–50 mm) on intertidal artificial structures*; *Create small ridges or ledges (1–50 mm) on intertidal artificial structures*; *Create large ridges or ledges (>50 mm) on intertidal artificial structures*; *Create groove habitats* and *small protrusions, ridges or ledges (1–50 mm) on intertidal artificial structures*.

Dafforn K.A. (2017) Eco-engineering and management strategies for marine infrastructures to reduce establishment and dispersal of non-indigenous species. *Management of Biological Invasions*, 8, 153–161.

Guichard F. & Bourget E. (1998) Topgraphic heterogeneity, hydrodynamics, and benthic community structure: a scale-dependence cascade. *Marine Ecology Progress Series*, 171, 59–70.

Harmelin-Vivien M.L., Harmelin J.G. & Leboulleux V. (1995) Microhabitat requirements for settlement of juvenile sparid fishes on Mediterranean rocky shores. *Hydrobiologia*, 300, 309–320.

MacArthur M., Naylor L.A., Hansom J.D. & Burrows M.T. (2020) Ecological enhancement of coastal engineering structures: passive enhancement techniques. *Science of the Total Environment*, 740, 139981.

Strain E.M.A., Olabarria C., Mayer-Pinto M., Cumbo V., Morris R.L., Bugnot A.B., Dafforn K.A., Heery E., Firth L.B., Brooks P.R. & Bishop M.J. (2018) Eco-engineering urban infrastructure for marine and coastal biodiversity: which interventions have the greatest ecological benefit? *Journal of Applied Ecology*, 55, 426–441.

Williams G.A. & Morritt D. (1995) Habitat partitioning and thermal tolerance in a tropical limpet, *Cellana grata*. *Marine Ecology Progress Series*, 124, 89–103.

A replicated, controlled study in 2015–2017 on an intertidal seawall on open coastline in the UK (1) found that boulders positioned with large protrusions on their upper surfaces, along with large ridges, supported similar macroalgae and invertebrate species richness and barnacle *Semibalanus balanoides* abundance, but higher limpet *Patella vulgata* abundance, than boulders positioned randomly. Boulders positioned with large protrusions and ridges on their upper surfaces supported similar macroalgae and invertebrate species richness (4 species/boulder) and barnacle abundance (data not reported) but more limpets (82 limpets/boulder) than boulders positioned randomly (4 species/boulder, 27 limpets/boulder). It is not clear whether these effects were the direct result of creating large protrusions or ridges. Ten granite boulders (width: 2 m) were intentionally positioned with naturally-occurring large protrusions and/or ridges on their upper surfaces (average 4/boulder) and ten were positioned randomly (1/boulder) at mid-highshore in a granite boulder seawall during construction in 2015–2017. Protrusions/ridges were 100–800 mm high (other dimensions/spacing not reported). Macroalgae and invertebrates on the upper surfaces of boulders were counted during low tide in June 2017.

A replicated, randomized, controlled study in 2014–2016 on an intertidal seawall in a marina in the Coral Sea, Australia (2) reported that large protrusions created on the seawall supported similar macroalgae and invertebrate species richness to seawall surfaces without protrusions, but that tilted protrusions with shaded surfaces supported different community composition to horizontal ones. Over 24 months, a total of nine macroalgae and invertebrate species groups were recorded on landward-tilted protrusions, eight on seaward-tilted protrusions, eight on horizontal protrusions, and 10 on seawall surfaces without protrusions (data not statistically tested). Community composition was similar on landward- and seaward-tilted protrusions, but both differed to horizontal protrusions (data reported as statistical model results). Four species (3 mobile invertebrates, 1 non-mobile invertebrate) recorded on protrusions were absent from seawall surfaces without. Large protrusions were created by attaching concrete troughs to a boulder seawall in June 2014. Troughs contained rock pools but outside surfaces were surveyed separately and constituted protrusions lacking a top surface. Rectangular protrusions (length: 400 mm; width: 250 mm; height: 350 mm) were either horizontal or tilted 45° towards the land or sea. Underhanging surfaces of tilted protrusions were shaded. There were three of each randomly arranged at midshore along the seawall. Macroalgae and invertebrates were counted on protrusions and seawall surfaces (number/dimensions not reported) during low tide over 24 months. One horizontal protrusion was missing and no longer provided habitat.

(1) MacArthur M., Naylor L.A., Hansom J.D. & Burrows M.T. (2020) Ecological enhancement of coastal engineering structures: passive enhancement techniques. *Science of the Total Environment*, 740, 139981.
(2) Waltham N.J. & Sheaves M. (2018) Eco-engineering rock pools to a seawall in a tropical estuary: microhabitat features and fine sediment accumulation. *Ecological Engineering*, 120, 631–636.

2.13. Create small ridges or ledges (1–50 mm) on intertidal artificial structures

- **Four studies** examined the effects of creating small ridges or ledges on intertidal artificial structures on the biodiversity of those structures. Two studies were on island coastlines in the Singapore Strait[1a,1b] and two were in estuaries in Hong Kong[2] and southeast Australia[3].

COMMUNITY RESPONSE (4 STUDIES)

- **Overall community composition (2 studies):** One of two replicated, randomized, controlled studies in Singapore[1a,1b] found that creating small ridges on intertidal artificial structures did not alter the combined macroalgae and invertebrate community composition on structure surfaces[1a]. One study[1b] found that creating small ridges, along with grooves, small protrusions and pits, had mixed effects on the community composition, depending on the site, and the size and arrangement of ridges and other habitats.
- **Overall richness/diversity (4 studies):** One of two replicated, randomized, controlled studies in Singapore[1a,1b] found that creating small ridges on intertidal artificial structures did not increase the combined macroalgae and invertebrate species richness on structure surfaces[1a]. One study[1b] found that creating small ridges, along with grooves, small protrusions and pits, did increase the

species richness, and that varying the habitat size and arrangement had mixed effects, depending on the shore level. Two replicated studies (including one randomized, paired sites study) in Hong Kong[2] and Australia[3] found that small ridges[2] or ledges[3] supported lower species richness than grooves created in between them, but one of them[2] found that species diversity on ridges compared with grooves varied depending on the ridge height.

- **Invertebrate richness/diversity (1 study):** One replicated study in Australia[3] found that small ledges created on intertidal artificial structures supported lower mobile invertebrate species richness than grooves created in between them.
- **Fish richness/diversity (1 study):** One replicated study in Australia[3] found that small ledges created on intertidal artificial structures supported similar fish species richness to grooves created in between them.

POPULATION RESPONSE (3 STUDIES)

- **Overall abundance (3 studies):** One of two replicated, randomized, controlled studies in Singapore[1a,1b] found that creating small ridges on intertidal artificial structures did not increase the combined macroalgae and invertebrate abundance on structure surfaces[1a]. One study[1b] found that creating small ridges, along with grooves, small protrusions and pits, had mixed effects on abundance, depending on the shore level, site, and the size and arrangement of ridges and other habitats. One replicated study in Australia[3] found that small ledges supported similar abundance to grooves created in between them.
- **Invertebrate abundance (1 study):** One replicated study in Australia[3] found that small ledges created on intertidal artificial structures supported lower mobile invertebrate and oyster abundances than grooves created in between them.
- **Fish abundance (1 study):** One replicated study in Australia[3] found that small ledges created on intertidal artificial structures supported similar fish abundance to grooves created in between them.

BEHAVIOUR (0 STUDIES)

Background

Small ridges and ledges create vertical or horizontal (i.e. overhangs) relief in intertidal rocky habitats. They can provide organisms refuge from desiccation and temperature fluctuations during low tide (Williams & Morrit 1995) and also shelter from predation or grazing (Wahl & Hoppe 2002). Some species preferentially recruit to habitats with high vertical or horizontal relief (Harmelin-Vivien *et al.* 1995). The size and density of ridges and ledges is likely to affect the size, abundance and variety of organisms that can use them. Small habitats can provide refuge for small-bodied organisms but may exclude larger organisms and limit their growth. Large habitats can be used by larger-bodied organisms but may not provide sufficient refuge from predators for smaller organisms. By default, horizontal ledges (overhangs) create shaded and downward-facing surfaces, which can be associated with the presence of non-native species (Dafforn 2017).

Ridges and ledges are sometimes present on quarried boulders used in marine artificial structures (MacArthur *et al.* 2020) but are often absent from other types of structures. Small ridges and ledges can be created on intertidal artificial structures by adding material, either during construction or retrospectively.

Definition: 'Small ridges and ledges' are elevations with a length to width ratio >3:1 that protrude 1–50 mm from the substratum (modified from "Small elevations" in Strain *et al.* 2018). On vertical surfaces, vertically-orientated elevations that fit these criteria are referred to as 'ridges', while horizontal ones are referred to as 'ledges'. On horizontal surfaces, these features are referred to as 'ridges' regardless of their orientation.

See also: *Create textured surfaces (≤1 mm) on intertidal artificial structures*; *Create natural rocky reef topography on intertidal artificial structures*; *Create small protrusions (1–50 mm) on intertidal artificial structures*; *Create large protrusions (>50 mm) on intertidal artificial structures*; *Create large ridges or ledges (>50 mm) on intertidal artificial structures*; *Create groove habitats* and *small protrusions, ridges or ledges (1–50 mm) on intertidal artificial structures.*

Dafforn K.A. (2017) Eco-engineering and management strategies for marine infrastructures to reduce establishment and dispersal of non-indigenous species. *Management of Biological Invasions*, 8, 153–161.

Harmelin-Vivien M.L., Harmelin J.G. & Leboulleux V. (1995) Microhabitat requirements for settlement of juvenile sparid fishes on Mediterranean rocky shores. *Hydrobiologia*, 300, 309–320.

MacArthur M., Naylor L.A., Hansom J.D. & Burrows M.T. (2020) Ecological enhancement of coastal engineering structures: passive enhancement techniques. *Science of the Total Environment*, 740, 139981.

Strain E.M.A., Olabarria C., Mayer-Pinto M., Cumbo V., Morris R.L., Bugnot A.B., Dafforn K.A., Heery E., Firth L.B., Brooks P.R. & Bishop M.J. (2018) Eco-engineering urban infrastructure for marine and coastal biodiversity: which interventions have the greatest ecological benefit? *Journal of Applied Ecology*, 55, 426–441.

Wahl M. & Hoppe K. (2002) Interactions between substratum rugosity, colonization density and periwinkle grazing efficiency. *Marine Ecology Progress Series*, 225, 239–249.

Williams G.A. & Morritt D. (1995) Habitat partitioning and thermal tolerance in a tropical limpet, *Cellana grata*. *Marine Ecology Progress Series*, 124, 89–103.

A replicated, randomized, controlled study in 2009–2010 on two intertidal seawalls on island coastlines in the Singapore Strait, Singapore (1a; same experimental set-up as 1b) found that concrete settlement plates with small ridges supported similar macroalgae and invertebrate community composition, species richness and abundance to granite plates without ridges. After 13 months, macroalgae and invertebrate community composition (data reported as statistical model results), species richness and abundance were statistically similar on settlement plates with small ridges (6 species/plate; 97 individuals/plate) and without (3 species/plate, 178 individuals/plate). Settlement plates (200 × 200 mm) were moulded with and without small ridges. Plates with ridges were concrete with 12 serrated ridges/plate, with either regular (16 mm width, height and spacing) or variable (4–28 mm) arrangement. Plates without ridges were granite fragments set in cement. Granite may be considered an environmentally-sensitive material compared with concrete (see "*Use environmentally-sensitive material on* *intertidal artificial structures*"). Eight of each design were randomly arranged at both lowshore and highshore on each of two granite boulder seawalls in November–December 2009. Macroalgae on plates were counted from photographs and invertebrates in the laboratory after 13 months.

A replicated, randomized, controlled study in 2009–2010 on two intertidal seawalls on island coastlines in the Singapore Strait, Singapore (1b; same experimental set-up as 1a) found that concrete settlement plates with small ridges, along with grooves, small protrusions and pits, supported higher macroalgae and invertebrate species richness than granite plates without added habitats, but that abundances and community composition varied depending on the habitat arrangement, shore level and site. After 13 months, macroalgae and invertebrate species richness was higher on settlement plates with ridges, grooves, protrusions and pits than without at lowshore (13–23 vs 6–10 species/plate) and highshore (5–9 vs 2–3/plate). Richness was higher on plates with variable habitats than regular ones at lowshore (22–23 vs 13–16/plate), but not highshore (6–9 vs 5–6/plate). Abundances were higher on plates with added habitats than without in four of eight comparisons (9–833 vs 3–208 individuals/plate), while community composition differed in three of four comparisons (data reported as statistical model results). In all other comparisons, results were similar (abundances: 104–1,957 vs 49–1,162/plate). It is not clear whether these effects were the direct result of creating ridges, grooves, protrusions or pits. However, richness was lower on plate quarters with ridges (6 species/plate) than pits (11/plate), but similar to quarters with protrusions and with grooves *and* ridges (both 8/plate). Abundances were similar for all habitat types (88–231 individuals/quarter). Settlement plates (400 × 400 mm) were moulded with and without small ridges, along with grooves, small protrusions and pits. Plates with added habitats were concrete. Each 200 × 200 mm quarter contained either 12 serrated ridges, four-to-five grooves and ridges, 36 protrusions or 36 pits. All habitats had either regular (16 mm width, depth/height and spacing) or variable (4–28 mm) arrangement. Plates without added habitats were granite fragments set in cement. Granite may be considered an environmentally-sensitive material compared with concrete (see "*Use environmentally-sensitive material on intertidal artificial structures*"). Eight of each design were randomly arranged at both lowshore and highshore on each of two granite boulder seawalls in November–December 2009. Macroalgae on plates were counted from photographs and invertebrates in the laboratory after 13 months.

A replicated, randomized, paired sites study in 2016–2017 on two intertidal seawalls in the Pearl River estuary, Hong Kong (2) found that small ridges created on the seawalls supported lower macroalgae and invertebrate species richness than groove habitats created in between them, while species diversity varied depending on the ridge height. After 12 months, macroalgae and invertebrate species richness on settlement plates was similar on tall and short ridges (both 3–4 species/plate), and lower on both than in the grooves in between (8–9/plate). The same was true for species diversity, except that tall ridges supported similar diversity to grooves (data reported as Shannon index). Concrete settlement plates (250 × 250 mm) were moulded with five tall (height: 50 mm) or short (25 mm) vertical small ridges (length: 250 mm; width: 17–65 mm) between four grooves (length: 250 mm; width: 15–50 mm; depth: 50 or 25 mm). Five of each were randomly arranged at midshore on each of two vertical concrete seawalls in November 2016 (month/year: M. Perkins *pers. comms.*). Plates had textured surfaces. Macroalgae and invertebrates on ridges and in grooves were counted in the laboratory after 12 months. One plate with tall ridges was missing and no longer provided habitat.

A replicated study in 2015–2016 of two intertidal seawalls in Sydney Harbour estuary, Australia (3) found that small ledges created on the seawalls supported lower macroalgae and invertebrate species richness and lower mobile invertebrate and oyster *Saccostrea glomerata* abundances than groove habitats created in between them, but that macroalgae and other non-mobile invertebrate abundance, fish species richness and fish abundance were similar on ledges and in grooves. After 12 months, small ledges supported lower macroalgae and non-mobile invertebrate species richness (4 species/ledge) than the grooves in between (6/groove). The same was true for mobile invertebrates (4/ledge vs 6/groove), but not fishes (both 2/sample). Abundances were lower on ledges than in grooves for mobile invertebrates (1 individual/ledge vs 18/groove) and oysters (14 vs 56% cover), but were similar for macroalgae and other non-mobile invertebrates (36 vs 38% cover) and fishes (both 1 individual/sample). Concrete settlement plates (250 × 250 mm) were moulded with five horizontal small ledges (length: 250 mm; width: 17–65 mm; height: 50 mm) between four grooves (length: 250 mm; width: 15–50 mm; depth: 50 mm). Five plates were attached at midshore on each of two vertical sandstone seawalls in November 2015. Plates had textured surfaces. Macroalgae and invertebrates were counted on ledges and in grooves during low tide, from photographs and in the laboratory after 12 months. Fishes were counted from time-lapsed photographs during two high tides.

(1) Loke L.H.L. & Todd P.A. (2016) Structural complexity and component type increase intertidal biodiversity independently of area. *Ecology*, 97, 383–393.
(2) Bradford T.E., Astudillo J.C., Lau E.T.C., Perkins M.J., Lo C.C., Li T.C.H., Lam C.S., Ng T.P.T., Strain E.M.A., Steinberg P.D. & Leung K.M.Y. (2020) Provision of refugia and seeding with native bivalves can enhance biodiversity on vertical seawalls. *Marine Pollution Bulletin*, 160, 111578.
(3) Strain E.M.A., Cumbo V.R., Morris R.L., Steinberg P.D. & Bishop M.J. (2020) Interacting effects of habitat structure and seeding with oysters on the intertidal biodiversity of seawalls. *PLoS ONE*, 15, e0230807.

2.14. Create large ridges or ledges (>50 mm) on intertidal artificial structures

- **Three studies** examined the effects of creating large ridges or ledges on intertidal artificial structures on the biodiversity of those structures. Two studies were in an estuarine sound in northwest USA[1,3] and one was on an open coastline in the UK[2].

COMMUNITY RESPONSE (2 STUDIES)

- **Overall richness/diversity (2 studies):** One of two replicated, controlled studies (including one randomized study) in the USA[1] and the UK[2] reported that creating large ledges on intertidal artificial structures, along with grooves and small protrusions, increased the combined macroalgae, microalgae and invertebrate species diversity on structure surfaces. One study[2] found that creating large ridges, along with large protrusions, did not increase the combined macroalgae and invertebrate species richness.

POPULATION RESPONSE (3 STUDIES)

- **Overall abundance (1 study):** One replicated, randomized, controlled study in the USA[1] reported that creating large ledges on intertidal artificial structures, along with grooves and small protrusions, increased the combined macroalgae, microalgae and invertebrate abundance on structure surfaces.
- **Algal abundance (1 study):** One replicated, randomized, controlled study in the USA[1] found that creating large ledges on intertidal artificial structures, along with grooves and small protrusions, increased the rockweed abundance on structure surfaces.
- **Invertebrate abundance (2 studies):** Two replicated, controlled studies (including one randomized study) in the USA[1] and the UK[2] found that creating large ledges[1] or ridges[2] on intertidal artificial structures, along with grooves and small protrusions[1], or large protrusions[2], increased the abundance of mussels[1] or limpets[2], but not barnacles[2], on structure surfaces.
- **Fish abundance (1 study):** One before-and-after study in the USA[3] reported that creating large ledges on an intertidal artificial structure, along with grooves and small protrusions, did not increase juvenile salmon abundance around the structure.

BEHAVIOUR (1 STUDY)

- **Fish behaviour change (1 study):** One before-and-after study in the USA[3] reported that creating large ledges on an intertidal artificial structure, along with grooves and small protrusions, increased juvenile salmon feeding activity around the wall.

Background

Large ridges and ledges create vertical or horizontal (i.e. overhangs) relief in intertidal rocky habitats. They can provide organisms refuge from desiccation and temperature fluctuations during low tide (Williams & Morrit 1995) and alter flow velocities (Guichard & Bourget 1998). Some species preferentially recruit to habitats with high vertical or horizontal relief, potentially to avoid predators (Harmelin-Vivien *et al.* 1995). The size and density of ridges and ledges is likely to affect the size, abundance and variety of organisms that can use them. Small habitats can provide refuge for small-bodied organisms but may exclude larger organisms and limit their growth. Large habitats can be used by larger-bodied organisms but may not provide sufficient refuge from predators for smaller organisms. By default, horizontal ledges (overhangs) create shaded and downward-facing surfaces, which can be associated with the presence of non-native species (Dafforn 2017).

Ridges and ledges are sometimes present on quarried boulders used in marine artificial structures (MacArthur *et al.* 2020) but are often absent from other types of structures. Large ridges and ledges can be created on intertidal artificial structures by adding material, either during construction or retrospectively.

Definition: 'Large ridges and ledges' are elevations with a length to width ratio >3:1 that protrude >50 mm from the substratum (modified from "Large elevations" in Strain *et al.* 2018). On vertical surfaces, vertically-orientated elevations that fit these criteria are referred to as 'ridges', while horizontal ones are referred to as 'ledges'. On horizontal surfaces, these features are referred to as 'ridges' regardless of their orientation.

See also: *Create textured surfaces (≤1 mm) on intertidal artificial structures*; *Create natural rocky reef topography on intertidal artificial structures*; *Create small protrusions (1–50 mm) on intertidal artificial structures*; *Create large protrusions (>50 mm) on intertidal artificial structures*; *Create small ridges or ledges (1–50 mm) on intertidal artificial structures*; *Create groove habitats* and *small protrusions, ridges or ledges (1–50 mm) on intertidal artificial structures*.

Dafforn K.A. (2017) Eco-engineering and management strategies for marine infrastructures to reduce establishment and dispersal of non-indigenous species. *Management of Biological Invasions*, 8, 153–161.

Guichard F. & Bourget E. (1998) Topgraphic heterogeneity, hydrodynamics, and benthic community structure: a scale-dependence cascade. *Marine Ecology Progress Series*, 171, 59–70.

Harmelin-Vivien M.L., Harmelin J.G. & Leboulleux V. (1995) Microhabitat requirements for settlement of juvenile sparid fishes on Mediterranean rocky shores. *Hydrobiologia*, 300, 309–320.

MacArthur M., Naylor L.A., Hansom J.D. & Burrows M.T. (2020) Ecological enhancement of coastal engineering structures: passive enhancement techniques. *Science of the Total Environment*, 740, 139981.

Strain E.M.A., Olabarria C., Mayer-Pinto M., Cumbo V., Morris R.L., Bugnot A.B., Dafforn K.A., Heery E., Firth L.B., Brooks P.R. & Bishop M.J. (2018) Eco-engineering urban infrastructure for marine and coastal biodiversity: which interventions have the greatest ecological benefit? *Journal of Applied Ecology*, 55, 426–441.

Williams G.A. & Morritt D. (1995) Habitat partitioning and thermal tolerance in a tropical limpet, *Cellana grata*. *Marine Ecology Progress Series*, 124, 89–103.

A replicated, randomized, controlled study in 2008–2011 on three intertidal seawalls in Puget Sound estuary, USA (1) reported that large ledges created on seawall panels, along with grooves and small protrusions, supported higher macroalgae, microalgae and invertebrate species diversity and live cover, with more rockweed *Fucus distichus* and mussels *Mytilus* spp., than seawall surfaces without added habitats. After 42 months, the macroalgae, microalgae and invertebrate species diversity was higher on seawall panels with ledges, grooves and protrusions than on seawall surfaces without (data reported as Evenness index, not statistically tested). Total live cover was 83–84% on panels with ledges, grooves and protrusions and 74% on surfaces without (data not statistically tested). Rockweed and mussel abundances were statistically similar on panels with long ledges (rockweed: 5% cover; mussels: 6%) and short ledges (rockweed: 13%; mussels: 12%), and higher on both than on seawall surfaces without (both 1%). Abundances of six other species groups were not statistically tested (see paper for results). It is not clear whether these effects were the direct result of creating ledges, grooves or protrusions. Large ledges were created on concrete seawall panels (height: 2.3 m; width: 1.5 m; thickness: ~150 mm) using a formliner. Each panel had three long (length: ~1.5 m; width/height: ~0.5 m) or six short (length: ~0.7 m; width: ~0.2 m; height: ~0.5 m) evenly-spaced horizontal ledges. Panels were either smooth or had grooves and small protrusions on their surfaces. One panel of each ledge-surface combination was randomly arranged spanning high–lowshore on each of three vertical concrete seawalls in January 2008. Seawall surfaces were intertidal areas of seawall cleared of organisms (dimensions/spacing not reported). Macroalgae, microalgae and invertebrates were counted on panels (excluding downward-facing surfaces) and seawall surfaces during low tide after 42 months.

A replicated, controlled study in 2015–2017 on an intertidal seawall on open coastline in the UK (2) found that boulders positioned with large ridges on their upper surfaces, along with large protrusions, supported similar macroalgae and invertebrate species richness and barnacle *Semibalanus balanoides* abundance, but higher limpet *Patella vulgata* abundance, than boulders positioned randomly. Boulders positioned with large ridges and protrusions on their upper surfaces supported similar macroalgae and invertebrate species richness (4 species/boulder) and barnacle abundance (data not reported) but more limpets (82 limpets/boulder) than boulders positioned randomly (4 species/boulder, 27 limpets/boulder). It is not clear whether these effects were the direct result of creating large ridges or protrusions. Ten granite boulders (width: 2 m) were intentionally positioned with naturally-occurring large ridges and/or protrusions on their upper surfaces (average 4/boulder) and ten were positioned randomly (1/boulder) at mid-highshore in a granite boulder seawall during construction in 2015–2017. Ridges/protrusions were 100–800 mm high (other dimensions/spacing not reported). Macroalgae and invertebrates on the upper surfaces of boulders were counted during low tide in June 2017.

A before-and-after study in 2012–2018 on an intertidal seawall in Puget Sound estuary, USA (3) reported that creating large ledges on the seawall, along with grooves and small protrusions, did not increase juvenile salmon *Oncorhynchus* spp. abundance around the wall but increased their feeding activity. Data were not statistically tested. Juvenile salmon abundances were lower after large ledges were created during seawall reconstruction (5–151 individuals/m^2) compared with before (47–431/100m^2), but the frequency of their feeding behaviour increased by 6–27%. It is not clear whether these effects were the direct result of creating ledges, grooves and protrusions, increased light levels or reduced water depth in front of the wall. Large ledges (length: 2 m; width: 0.6 m; height: 0.2 m) were created on concrete seawall panels using a formliner. Each panel had one horizontal ledge at high, mid or lowshore and grooves and small protrusions on their surfaces. Panels were attached to a vertical concrete seawall during reconstruction in 2017 (numbers/month not reported). Light-penetrating panels were also installed to increase light around the wall, and the seabed was raised in front. Juvenile salmon within 10 m of the wall were surveyed from 20–minute snorkels at high and low tide during March–August at three sites along the wall before reconstruction in 2012 (35 surveys), and at three different sites along the wall after reconstruction in 2018 (42 surveys).

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M.K. La Peyre & J.D. Toft (eds.) *Living Shorelines: The Science And Management Of Nature-Based Coastal Protection*. CRC Press, Boca Raton, Florida.
(2) MacArthur M., Naylor L.A., Hansom J.D. & Burrows M.T. (2020) Ecological enhancement of coastal engineering structures: passive enhancement techniques. *Science of the Total Environment*, 740, 139981.
(3) Sawyer A.C., Toft J.D. & Cordell J.R. (2020) Seawall as salmon habitat: eco-engineering improves the distribution and foraging of juvenile Pacific salmon. *Ecological Engineering*, 151, 105856.

2.15. Create grooves *and* small protrusions, ridges or ledges (1–50 mm) on intertidal artificial structures

- **Sixteen studies** examined the effects of creating groove habitats *and* small protrusions, ridges or ledges on intertidal artificial structures on the biodiversity of those structures. Five studies were on island coastlines in the Singapore Strait[1,2a,2b,4,8], seven were in estuaries in northwest USA[3,12], southeast Australia[7,10,13,15] and Hong Kong[11], and one was in each of a marina in northern Israel[5] and a port in southeast Spain[6]. One was on an open coastline and in an estuary in the UK[9], and one was in 14 estuaries and bays worldwide[14].

COMMUNITY RESPONSE (14 STUDIES)

- **Overall community composition (9 studies):** Three of five replicated, randomized, controlled studies (including one paired sites, before-and-after study) in Singapore[2a,2b,4,8] and Israel[5] found that creating groove habitats and small ridges/ledges on intertidal artificial structures, along with holes and environmentally-sensitive material in one[5], altered the combined macroalgae and invertebrate community composition on structure surfaces[2a,4,5]. Two studies[2b,8] found that creating grooves and small ridges, along with pits in one[2b], had mixed effects on the community composition depending on the site[2b,8], the presence of water-retaining and light-shading covers[8], and the size and arrangement of grooves and ridges[2b]. In contrast, one of the studies[4] found that varying the size and arrangement had no significant effect. One of the studies[5], along with four other replicated, randomized, controlled studies in Singapore[1], Hong Kong[11] and Australia[13,15], reported that groove habitats and small ridges/ledges, along with pits[1] or holes and environmentally-sensitive material[5] in two studies, supported species that were absent from structure surfaces without grooves and ridges/ledges.
- **Overall richness/diversity (11 studies):** Six of 11 replicated, randomized, controlled studies (including one paired sites, before-and-after study) in Singapore[2a,2b,4,8], the USA[3], Israel[5], the UK[9], Hong Kong[11], Australia[13,15] and worldwide[14] found that creating groove habitats and small ridges/ledges on intertidal artificial structures, along with pits[2b] or holes and environmentally-sensitive material[5] in two studies, increased the combined macroalgae and invertebrate species diversity[5] and/or richness[2a,2b,4,5,8,13] on structure surfaces. Five studies found that creating grooves and small protrusions/ridges/ledges, along with large ledges[3] or using environmentally-sensitive material[9] in two, had mixed effects on species diversity[3,15] and/or richness[9,11,14,15], depending on the depth/height of grooves and ridges[11,14,15], the presence of large ledges on structure surfaces[3], the shore level[14], species group[14] and site[9,11,14,15]. One of the studies[4] found that varying the size and arrangement of grooves and ridges increased the species richness, while one[2b] found that effects depended on the shore level. One of the studies[8] found that partially-covering grooves and ridges with water-retaining and light-shading covers increased the species richness.

- **Algal richness/diversity (2 studies):** One of two replicated, randomized, controlled studies in Singapore[1] and worldwide[14] found that creating groove habitats and small ridges on intertidal artificial structures had mixed effects on the macroalgal species richness on structure surfaces, depending on the size of grooves and ridges and the location[14]. One study[1] found that creating grooves and ridges, along with pits, increased the species richness, regardless of their size and arrangement.
- **Invertebrate richness/diversity (2 studies):** One of two replicated, randomized, controlled studies in Australia[13] and worldwide[14] found that creating groove habitats and small ridges on intertidal artificial structures had mixed effects on the mobile and non-mobile invertebrate species richness on structure surfaces, depending on the size of grooves and ridges and the location[14]. One study[13] found that creating grooves and small ledges increased the mobile invertebrate species richness.
- **Fish richness/diversity (3 studies):** Two of three replicated, randomized, controlled studies in Australia[7,10,13] found that creating groove habitats and small ridges/ledges on intertidal artificial structures did not increase the fish species richness on and around structure surfaces[10,13]. One study[7] found that creating grooves and ridges had mixed effects on fish species richness depending on the site.

POPULATION RESPONSE (13 STUDIES)

- **Overall abundance (6 studies):** Two of six replicated, randomized, controlled studies in Singapore[2a,2b,8], the USA[3] and Australia[13,15] found that creating groove habitats and small ridges/ledges on intertidal artificial structures did not increase the combined macroalgae and invertebrate abundance on structure surfaces[2a,13]. Two studies[3,8] found that creating grooves and small protrusions/ridges, along with large ledges in one[3], and when partially-covered with water-retaining and light-shading covers in the other[8], did increase abundance. Two[2b,15] found that creating grooves and small ridges/ledges, along with pits in one[2b], had mixed effects on abundance depending on the size[2b,15] and arrangement[2b] of grooves and ridges/ledges, the shore level[2b] and/or the site[2b,15].
- **Algal abundance (3 studies):** Two of three replicated, randomized, controlled studies in Singapore[1], the USA[3] and worldwide[14] found that creating groove habitats and small protrusions/ridges on intertidal artificial structures, along with large ledges in one[3], had mixed effects on rockweed[3] or combined macroalgal[14] abundance, depending on the presence of large ledges on structure surfaces[3], the depth/height of grooves and ridges[14], the shore level[14] and/or the site[14]. One study[1] found that creating grooves and small ridges, along with pits, did not increase the macroalgal abundance, regardless of the size and arrangement of grooves and ridges.
- **Invertebrate abundance (7 studies):** Five of seven replicated, randomized, controlled studies in the USA[3], Singapore[8], the UK[9], Hong Kong[11], Australia[13,15] and worldwide[14] found that creating groove habitats and small protrusions/ridges/ledges on intertidal artificial structures, along with large ledges[3] or using environmentally-sensitive material[9] in two, had mixed effects on mobile invertebrate[9,11,14], non-mobile invertebrate[11,14], limpet[8], mussel[3], juvenile oyster[11] and/or barnacle[9,11] abundances, depending on the depth/height of grooves and ridges[11,14], the presence of large ledges[3] or water-retaining and light-shading covers[8], the shore level[14], and/or the

site[8,9,11,14]. Two studies[13,15] found that creating grooves and small ridges/ledges increased oyster but not mobile invertebrate abundance on structure surfaces.

- **Fish abundance (4 studies):** Three replicated, randomized, controlled studies and one before-and-after study in Australia[7,10,13] and the USA[12] found that creating groove habitats and small ridges/ledges on intertidal artificial structures, along with large ledges in one study[12], did not increase combined fish[7,10,13] or juvenile salmon[12] abundances on and around structure surfaces, regardless of whether there were transplanted oysters and/or algae on structure surfaces[10].

BEHAVIOUR (3 STUDIES)

- **Use (1 study):** One replicated study in Spain[6] reported that grooves and small protrusions created on an intertidal artificial structure were colonized by a number of microalgal species.
- **Fish behaviour change (2 studies):** One replicated, randomized, controlled study in Australia[10] found that creating groove habitats and small ledges on intertidal artificial structures increased the time benthic fishes spent interacting with structure surfaces but decreased the number of bites they took and did not change pelagic fish behaviour. One before-and-after study in the USA[12] reported that creating grooves and small protrusions, along with large ledges, increased juvenile salmon feeding activity around the structure.

Background

Grooves, small protrusions, ridges and ledges provide organisms refuge from desiccation and temperature fluctuations during low tide in intertidal rocky habitats (Williams & Morrit 1995). They also provide shelter from predation or grazing (Menge & Lubchenco 1981; Wahl & Hoppe 2002) and some species preferentially settle in and around them (Chabot & Bourget 1988; Harmelin-Vivien *et al.* 1995). The size and density of grooves, protrusions, ridges and ledges is likely to affect the size, abundance and variety of organisms that can use them. Small habitats can provide refuge for small-bodied organisms but may exclude larger organisms, limit their growth and get rapidly filled-up (Firth *et al.* 2020). Large habitats can be used by larger-bodied organisms but may not provide sufficient refuge from predators for smaller organisms. By default, horizontal protrusions/ledges (overhangs) create shaded and downward-facing surfaces, which can be associated with the presence of non-native species (Dafforn 2017).

Grooves, protrusions, ridges and ledges are sometimes present on quarried boulders used in marine artificial structures (MacArthur *et al.* 2020), but are absent from many other structures (Aguilera *et al.* 2014). Grooves can form on structures through erosion, but will often be filled or repaired during maintenance works (Moreira *et al.* 2007). Groove habitats, small protrusions, ridges and ledges can be created on intertidal artificial structures by adding or removing material, either during construction or retrospectively. In some scenarios, creating one will automatically result in creation of the other (i.e. grooves created in between created protrusions/ridges/ledges, or *vice versa*). Studies containing such scenarios are considered under this joint intervention.

Definition: 'Groove habitats' are depressions with a length to width ratio >3:1 and depth 1–50 mm (modified from "Crevices" in Strain *et al.* 2018). 'Small protrusions' are elevations with a length to width ratio ≤3:1 that protrude 1–50 mm from the substratum (modified from "Small elevations" in Strain *et al.* 2018). 'Small ridges and ledges' are elevations with a length to width ratio >3:1 that protrude 1–50 mm from the substratum (modified from "Small elevations" in Strain *et al.* 2018). On vertical surfaces, vertically-orientated elevations that fit these criteria are referred to as 'ridges', while horizontal ones are referred to as 'ledges'. On horizontal surfaces, these features are referred to as 'ridges' regardless of their orientation.

See also: *Create textured surfaces (≤1 mm) on intertidal artificial structures*; *Create natural rocky reef topography on intertidal artificial structures*; *Create pit habitats (1–50 mm) on intertidal artificial structures*; *Create hole habitats (>50 mm) on intertidal artificial structures*; *Create groove habitats (1–50 mm) on intertidal artificial structures*; *Create crevice habitats (>50 mm) on intertidal artificial structures*; *Create small protrusions (1–50 mm) on intertidal artificial structures*; *Create small ridges or ledges (1–50 mm) on intertidal artificial structures*.

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A replicated, randomized, controlled study in 2011–2012 on two intertidal seawalls on island coastlines in the Singapore Strait, Singapore (1; same experimental set-up as 4) reported that concrete settlement plates with groove habitats and small ridges, along with pits, supported higher macroalgal species richness but similar abundances compared with granite plates without added habitats. After 12 months, settlement plates with grooves, ridges and pits supported a total of five macroalgal species groups, while plates without supported three (data not statistically tested). Abundances of three species groups were statistically similar on plates with added habitats (18–41% cover) and without (5–61%) in five of six comparisons, while one group was more abundant on plates with added habitats (22–27 vs 5%) at one site. Abundances were similar on plates with variable (1–34%) and regular (3–41%) habitats. It is not clear whether these effects were the direct result of creating grooves, ridges or pits. Settlement plates (400 × 400 mm) were moulded with groove habitats and small ridges, with pits, and with neither. Plates with grooves, ridges and pits were concrete with four-to-five concentric circular grooves and ridges/plate or 36 pits/plate. Grooves, ridges and pits had either regular (32 mm width, depth/height and spacing) or variable (8–56 mm) arrangement. Plates without grooves, ridges or pits were granite fragments set in cement. Granite may be considered an environmentally-sensitive material compared with concrete (see "*Use environmentally-sensitive material on intertidal artificial structures*"). Five of each design were randomly arranged at lowshore on each of two granite boulder seawalls in July 2011. Macroalgae on plates were counted from photographs after 12 months.

A replicated, randomized, controlled study in 2009–2010 on two intertidal seawalls on island coastlines in the Singapore Strait, Singapore (2a; same experimental set-up as 2b) found that concrete settlement plates with groove habitats and small ridges supported different macroalgae and invertebrate community composition with higher species richness but similar abundances compared with granite plates without grooves and ridges. After 13 months, macroalgae and invertebrate species richness was higher on settlement plates with grooves and ridges (8 species/plate) than without (3/plate), while abundances were statistically similar (126 vs 178 individuals/plate). Community composition differed on plates with and without grooves and ridges (data reported as statistical model results). Settlement plates (200 × 200 mm) were moulded with and without groove habitats and small ridges. Plates with grooves and ridges were concrete with four-to-five concentric circular grooves and ridges/plate with either regular (16 mm width, depth/height and spacing) or variable (4–28 mm) arrangement. Plates without grooves and ridges were granite fragments set in cement. Granite may be considered an environmentally-sensitive material compared with concrete (see "*Use environmentally-sensitive material on intertidal artificial structures*"). Eight of each design were randomly arranged at both lowshore and highshore on each of two granite boulder seawalls in November–December 2009. Macroalgae on plates were counted from photographs and invertebrates in the laboratory after 13 months.

A replicated, randomized, controlled study in 2009–2010 on two intertidal seawalls on island coastlines in the Singapore Strait, Singapore (2b; same experimental set-up as 2a) found that concrete settlement plates with groove habitats and small ridges, along with small protrusions and pits, supported higher macroalgae and invertebrate species richness than granite plates without added habitats, while abundances and community composition varied depending on the habitat arrangement, shore level and site. After 13 months, macroalgae and invertebrate species richness was higher on settlement plates with grooves, ridges, protrusions and pits than without at lowshore (13–23 vs 6–10 species/plate) and highshore (5–9 vs 2–3/plate). Richness was higher on plates with variable habitats than regular ones at lowshore (22–23 vs 13–16/plate), but not highshore (6–9 vs 5–6/plate). Abundances were higher on plates with added habitats than without in four of eight comparisons (9–833 vs 3–208 individuals/plate), while community composition differed in three of four comparisons (data reported as statistical model results). In all other comparisons, results were similar (abundances: 104–1,957 vs 49–1,162/plate). It is not clear whether these effects were the direct result of creating grooves and ridges, protrusions or pits. However, plate quarters with grooves and ridges had similar richness (8 species/quarter) and abundances (126 individuals/quarter) to quarters with the other habitat types (6–11 species and 88–231 individuals/quarter). Settlement plates (400 × 400 mm) were moulded with and without groove habitats and small ridges, along with small protrusions and pits. Plates with added habitats were concrete. Each 200 × 200 mm quarter contained either four-to-five concentric circular grooves and ridges, 36 protrusions, 12 ridges or 36 pits. All habitats had either regular (16 mm width, depth/height and spacing) or variable (4–28 mm) arrangement. Plates without added habitats were granite fragments set in cement. Granite may be considered an environmentally-sensitive material compared with concrete (see "*Use environmentally-sensitive material on intertidal artificial structures*"). Eight of each design were randomly arranged at both lowshore and highshore on each of two granite boulder seawalls in November–December 2009. Macroalgae on plates were counted from photographs and invertebrates in the laboratory after 13 months.

A replicated, randomized, controlled study in 2008–2011 on three intertidal seawalls in Puget Sound estuary, USA (3) reported that seawall panels with grooves and small protrusions, along with large ledges, supported higher macroalgae, microalgae and invertebrate species diversity and live cover, with more rockweed *Fucus distichus* and mussels *Mytilus* spp., than seawall surfaces without added habitats, but that flat panels (i.e. without large ledges) with grooves and protrusions did not. After 42 months, ledged seawall panels with grooves and small protrusions supported higher macroalgae, microalgae and invertebrate species diversity (data reported as Evenness index, not statistically tested), rockweed abundance (5–13% cover) and mussel abundance (6–12%) than seawall surfaces without added habitats (rockweed/mussels: both 1%), but flat panels with grooves and potrusions did not (rockweed: 0%; mussels: 6%). Total live cover was 83–84% on ledged panels with grooves and protrusions, 81% on flat panels with grooves and protrusions, and 74% on seawall surfaces (data not statistically tested). Abundances of six other species groups were not statistically tested (see paper for results). It is not clear whether these effects were the direct result of creating grooves and protrusions or ledges. Concrete seawall panels (height: 2.3 m; width: 1.5 m; thickness: ~150 mm) were moulded with and without groove habitats and small protrusions (dimensions not reported) using a cobble-effect formliner. Panels had three long or six short horizontal large ledges, or were flat. One panel of each surface-ledge combination was randomly arranged, spanning high–lowshore on each of three vertical concrete seawalls in January 2008. Seawall surfaces were intertidal areas of seawall cleared of organisms (dimensions/spacing not reported). Macroalgae, microalgae and invertebrates were counted on panels (excluding downward-facing surfaces) and seawall surfaces during low tide after 42 months.

A replicated, randomized, controlled study in 2011–2012 on two intertidal seawalls on island coastlines in the Singapore Strait, Singapore (4; same experimental set-up as 1) found that concrete settlement plates with groove habitats and small ridges supported higher macroalgae and invertebrate species richness and different community composition compared with granite plates without grooves or ridges. After 12 months, settlement plates with variable grooves and ridges supported a total of 49 macroalgae and invertebrate species, while plates with regular grooves and ridges supported 35 species and plates without grooves and ridges supported 22 (data not statistically tested). Average richness was higher on plates with variable grooves and ridges (18 species/plate) than regular ones (13/plate), and higher on both than on plates without (7/plate). Community composition was similar on plates with variable and regular grooves and ridges, but both differed to plates without (data reported as statistical model results). Settlement plates (400 × 400 mm) were moulded with and without groove habitats and small ridges. Plates with grooves and ridges were concrete with four-to-five concentric circular grooves and ridges/plate with either regular (32 mm width, depth/height and spacing) or variable (8–56 mm) arrangement. Plates without grooves and ridges were granite fragments set in cement. Granite may be considered an environmentally-sensitive material compared with concrete (see "*Use environmentally-sensitive material on intertidal artificial structures*"). Five of each design were randomly arranged at lowshore on each of two granite boulder seawalls in July 2011. Macroalgae and invertebrates on plates were counted in the laboratory after 12 months.

A replicated, randomized, paired sites, controlled, before-and-after study in 2014–2016 on an intertidal seawall in a marina in the Mediterranean Sea, Israel (5) found that groove habitats and small ledges created on seawall panels, along with holes and environmentally-sensitive material, supported higher macroalgae and invertebrate species diversity and richness and different community composition compared with standard-concrete seawall surfaces without added habitats. After 22 months, macroalgae and invertebrate species diversity (data reported as Shannon index) and richness was higher on panels with added habitats (8 species/quadrat) than on seawall surfaces without (3/quadrat), and compared with seawall surfaces before habitats were added (2/quadrat). Community composition differed between panels with added habitats and seawall surfaces without (data reported as statistical model results). Five species groups (1 macroalgae, 4 non-mobile invertebrates) recorded on panels were absent from surfaces without. It is not clear whether these effects were the direct result of creating grooves and ledges, holes, or using environmentally-sensitive material. Groove habitats and small ledges were created on seawall panels (height: 1.5 m; width: 0.9 m; thickness: 130 mm) using a formliner. Each panel had multiple interlocking grooves and ledges (length: 50–150 mm; width/depth/height: 10–50 mm) amongst multiple holes. Panels were made from patented ECOncrete™ material. Four panels were attached to a vertical concrete seawall in November 2014. The top 0.3 m were intertidal. Panels were compared with standard-concrete seawall surfaces cleared of organisms (height: 0.3 m;

width: 0.9 m) adjacent to each panel. Macroalgae and invertebrates were counted in one 300 × 300 mm randomly-placed quadrat on each panel and seawall surface during high tide over 22 months.

A replicated study (year not reported) on an intertidal seawall in Ceuta Port in the Alboran Sea, Spain (6) reported that settlement plates with groove habitats and small protrusions had chlorophyll-a and 15 diatom species on their surfaces. After two months, chlorophyll-a density on plates with grooves and protrusions ranged from 3–18 μg/cm^2. Total abundances of 15 diatom species ranged from 1–752 individuals across all plates. Settlement plates (170 × 170 mm) were cut to create a regular grid of six groove habitats (length: 170 mm; width/depth: ~7 mm) between 16 square protrusions (length/width: 30 mm; height: ~7 mm) on their surfaces. Plates were either sandstone, limestone, gabbro, slate or concrete. One of each material was randomly arranged, horizontally, on each of five midshore boulders along a limestone boulder seawall (month/year not reported). Microalgae and chlorophyll-a on plates were measured using a scanning microscope and spectrophotometer, respectively, after two months.

A replicated, randomized, controlled study in 2015 on two intertidal seawalls in Sydney Harbour estuary, Australia (7) found that creating groove habitats and small ridges on settlement plates increased the species richness of fish on and around plates at one of two sites, but did not increase fish abundances. After one month, at one site, fish species richness was higher on and around settlement plates with grooves and ridges (7 species/plate) than without (4/plate), while at the second site, there was no difference (both 3/plate). Maximum fish abundance was similar on and around plates with and without grooves and ridges (2–5 vs 3–4 individuals/plate) at both sites. Concrete settlement plates (250 × 250 mm) were moulded with and without groove habitats and small ridges. Plates with grooves and ridges had four vertical grooves (length: 250 mm; width: 15–50 mm; depth: 50 mm) between five ridges (length: 250 mm; width: 17–65 mm; height: 50 mm). Five plates with grooves and ridges and five without were randomly arranged at midshore on each of two vertical sandstone seawalls in November 2015. Plates had textured surfaces and 52 juvenile oysters attached. Fishes were counted on and around plates from time-lapsed photographs during two high tides after one month.

A replicated, randomized, controlled study in 2010–2011 on two intertidal seawalls on island coastlines in the Singapore Strait, Singapore (8) found that concrete settlement plates with groove habitats and small ridges supported higher macroalgae and invertebrate species richness and abundance than granite plates without grooves or ridges, but that community composition and limpet *Siphonaria guamensis* abundance varied depending on the site and whether grooves and ridges were partially-covered. After eight months, macroalgae and invertebrate species richness and abundance were higher on settlement plates with partially-covered grooves and ridges (20 species/plate, 89 individuals/plate) than uncovered grooves and ridges (14 species/plate, 43 individuals/plate) and plates without grooves and ridges (10 species/plate, 40 individuals/plate). Community composition differed on plates with and without grooves and ridges in three of four comparisons (data reported as statistical model results). At one of two sites, there were 250 limpets/plate with partially-covered grooves and ridges, 420/plate with uncovered grooves and ridges and 225/plate without grooves and ridges (data not statistically tested). At the second site, limpet abundance was 0/plate for each. Settlement plates (200 × 200 mm) were moulded with and without groove habitats and small ridges. Plates with grooves and ridges were concrete with five concentric circular grooves and ridges/plate (8–56 mm width, depth/height and spacing). Some were partially-covered with water-retaining and light-shading plates. Plates without grooves or ridges were granite fragments set in cement. Granite may be considered an environmentally-sensitive material compared with concrete (see "*Use environmentally-sensitive material on intertidal artificial structures*"). Ten of each design were randomly arranged at lowshore on each of two granite boulder seawalls in August 2010. Macroalgae and invertebrates on plates were counted in the laboratory after eight months. Seven plates with grooves and ridges and four without were missing and no longer provided habitat.

A replicated, randomized, controlled study in 2016–2017 on two intertidal seawalls on open coastline in the English Channel and in the Forth estuary, UK (9) found that creating groove habitats and small ridges on settlement plates, along with using environmentally-sensitive material, increased the macroalgae and invertebrate species richness and invertebrate abundance on plates at one of two sites. After 18 months, at one of two sites, macroalgae and mobile invertebrate species richness was higher on plates with grooves and ridges (2 species/plate) than without (1/plate). The same was true for mobile invertebrate abundance (9 vs 1 individuals/plate) and barnacle (Cirripedia) cover (48 vs 34%). At the second site, plates with and without grooves and ridges supported similar richness (both 1 species/plate), mobile invertebrate abundance (1 vs 3 individuals/plate) and barnacle cover (84 vs 83%). It is not clear whether these effects were the direct result of creating grooves and ridges or using environmentally-sensitive material. Settlement plates (150 × 150 mm) were moulded with and without groove habitats and small ridges. Plates with grooves and ridges had six chevron-shaped grooves between seven ridges with variable dimensions (maximum depth/height: 30 mm). Eight limestone-cement (environmentally-sensitive material) plates with grooves and ridges and eight concrete plates without were randomly arranged at upper-midshore on each of two vertical concrete seawalls in April–May 2016. Macroalgae and invertebrates on plates were counted from photographs over 18 months.

A replicated, randomized, controlled study in 2016–2017 on three intertidal seawalls in Sydney Harbour estuary, Australia (10) found that creating groove habitats and small ledges on settlement plates did not increase fish species richness or abundance or alter pelagic fish behaviour, but altered benthic fish behaviour on and around plates. After 8–12 months, fish species richness and abundance were similar on and around settlement plates with and without grooves and ledges (data not reported). Benthic fishes spent longer interacting with plates with grooves and ledges (30 minutes/60-minute survey) than without (17 minutes/survey), but took fewer bites from their surfaces (2 vs 8 bites/survey). There were no significant differences for pelagic fishes (2 vs 1 minutes/survey, 8 vs 13 bites/survey). Concrete settlement plates (250 × 250 mm) were moulded with and without groove habitats and small ledges. Plates with grooves and ledges had four horizontal grooves (length: 250 mm; width: 15–50 mm; depth: 50 mm) between five ledges (length: 250 mm; width: 17–65 mm; height: 50 mm). Plates had textured surfaces with or without juvenile oysters, coralline algae, or both attached. Nine plates of each grooves/ledges-transplant combination were randomly arranged at mid-lowshore on each of three vertical sandstone seawalls in March 2016. Fishes were counted on and around one of each plate design from 60-minute videos during each of three high tides after 8–12 months. The time fishes spent within 50 mm of plates and the number of bites they took was recorded.

A replicated, randomized, controlled study in 2016–2017 on two intertidal seawalls in the Pearl River estuary, Hong Kong (11) found that creating groove habitats and small ridges on the seawalls had mixed effects on macroalgae and invertebrate species richness, and invertebrate abundances and biomass, depending on the depth/height of grooves and ridges and the site. After 12 months, in two of four comparisons, settlement plates with grooves and ridges supported higher macroalgae and invertebrate species richness (12–13 species/plate) and non-mobile invertebrate abundance (38–42% cover) than plates without (9 species/plate; 17% cover). In three of four comparisons, plates with grooves and ridges supported higher mobile invertebrate abundance (45–81 individuals/plate) and barnacle (Cirripedia) and oyster (*Saccostrea cuccullata*) recruit biomass (5–14 g/plate) than plates without (15–19 individuals/plate; 1 g/plate). In all other comparisons, plates with and without grooves and ridges were similar (richness: 11–12 vs 9 species/plate; non-mobiles: 14–27 vs 13% cover; mobiles: 31 vs 19 individuals/plate; barnacles/oysters: 2 vs 1 g/plate). Two mobile invertebrate species recorded on plates with grooves and ridges were absent from those without. Concrete settlement plates (250 × 250 mm) were moulded with and without groove habitats and small ridges. Plates with grooves and ridges had four vertical grooves (length: 250 mm; width: 15–50 mm) between five ridges (length: 250 mm; width: 17–65 mm). Grooves and ridges were either deep/tall (depth/height: 50 mm) or shallow/short (25 mm). Five of each were randomly arranged at midshore on each of two vertical concrete seawalls in November 2016 (month/year: M. Perkins *pers. comms.*). Plates had textured surfaces. Macroalgae and invertebrates on plates were counted from photographs and in the laboratory, and barnacle and oyster recruit biomass (dry weight) was measured after 12 months. One plate with deep/tall grooves and ridges was missing and no longer provided habitat.

A before-and-after study in 2012–2018 on an intertidal seawall in Puget Sound estuary, USA (12) reported that creating grooves and small protrusions, along with large ledges, on the seawall did not increase juvenile salmon *Oncorhynchus* spp. abundance around the wall, but increased their feeding activity. Data were not statistically tested. Juvenile salmon abundances were lower after grooves and small protrusions were created during seawall reconstruction (5–151 individuals/m^2) compared with before (47–431/100m^2), but the frequency of their feeding behaviour increased by 6–27%. It is not clear whether these effects were the direct result of creating grooves and protrusions, ledges, increased light levels or reduced water depth in front of the wall. Groove habitats and small protrusions (dimensions not reported) were created on concrete seawall panels using a cobble-effect formliner. Panels also had one large ledge on their surfaces. Panels were attached to a vertical concrete seawall during reconstruction in 2017 (numbers/month not reported). Light-penetrating panels were also installed to increase light around the wall, and the seabed was raised in front. Juvenile salmon within 10 m of the wall were surveyed from 20–minute snorkels at high and low tide during March–August at three sites along the wall before reconstruction in 2012 (35 surveys), and at three different sites along the wall after reconstruction in 2018 (42 surveys).

A replicated, randomized, controlled study in 2015–2016 on two intertidal seawalls in Sydney Harbour estuary, Australia (13) found that creating groove habitats and small ledges on the seawalls increased the macroalgae and invertebrate species richness and oyster *Saccostrea glomerata* abundance on seawall surfaces, but did not increase abundances of macroalgae and other invertebrates, or the species richness and abundance of fishes. After 12 months, species richness was higher on settlement plates with grooves and ledges than without for macroalgae and non-mobile invertebrates (6 vs 2 species/plate) and mobile invertebrates (7 vs 4/plate), while there was no difference for fishes (both 2/plate). Oyster abundance was higher on plates with grooves and ledges (34% cover) than without (8%) but there were no significant differences in the abundances of macroalgae and other non-mobile invertebrates (46 vs 31% cover), mobile invertebrates (20 vs 16 individuals/plate) or fishes (both 1/plate). Eighteen species (5 macroalgae, 3 non-mobile invertebrates, 9 mobile invertebrates, 1 fish) recorded on plates with grooves and ledges were absent from those without. Concrete settlement plates (250 × 250 mm) were moulded with and without groove habitats and small ledges. Plates with grooves and ledges had four horizontal grooves (length: 250 mm; width: 15–50 mm; depth: 50 mm) between five ledges (length: 250 mm; width: 17–65 mm; height: 50 mm). Five plates with grooves and ledges and five without were randomly arranged at midshore on each of two vertical sandstone seawalls in November 2015. Plates had textured surfaces. Macroalgae and invertebrates were counted on plates during low tide, from photographs and in the laboratory after 12 months. Fishes were counted on and around plates from time-lapsed photographs during two high tides.

A replicated, randomized, controlled study in 2015–2017 on 27 intertidal seawalls and breakwaters in 14 estuaries and bays worldwide (14) found that creating groove habitats and small ridges on settlement plates had mixed effects on the macroalgae and invertebrate species richness and abundance on plates, depending on the depth/height of grooves and ridges, the location, shore level and species group. After 12 months, plates with deep/tall grooves and ridges supported higher macroalgae and invertebrate species richness (4–28 species/plate) than plates without grooves and ridges (2–12/plate) in 11 of 14 locations, while in three locations there was no significant difference (2–6 vs 2–8/plate). Plates with shallow/short grooves and ridges supported higher richness (6–19/plate) than plates without (3–12/plate) in seven of 14 locations, while in seven locations there was no significant difference (both 2–8/plate). Out of 28 comparisons each time, plates with grooves and ridges supported higher macroalgal richness than plates without in two comparisons, higher macroalgal abundance in four, higher non-mobile invertebrate richness in 16, higher non-mobile invertebrate abundance in 11, and higher mobile invertebrate richness and abundance in 13 comparisons each. In all other comparisons, plates with and without grooves and ridges were similar (data not reported). [Significance results reported from Tables S5a,b in original paper]. Concrete settlement plates (250 × 250 mm) were moulded with and without groove habitats and small ridges. Plates with grooves and ridges had four vertical grooves (length: 250 mm; width: 15–50 mm) between five ridges (length: 250 mm; width: 17–65 mm). Grooves and ridges were either deep/tall (depth/height: 50 mm) or shallow/short (25 mm). Five of each were randomly arranged at highshore, midshore or lowshore on each of two vertical seawalls/breakwaters in each of 14 estuaries/bays worldwide between November 2015–2016. Plates had textured surfaces. Macroalgae and invertebrates on plates were counted in the laboratory after 12 months.

A replicated, randomized, controlled study in 2015–2016 on two intertidal seawalls in Sydney Harbour estuary, Australia (15) found that creating groove habitats and small ridges or ledges on the seawalls had mixed effects on macroalgae and invertebrate species richness, diversity and abundances, depending on the depth/height of grooves and ridges, the species group and site. After 12 months, the macroalgae and invertebrate species richness was higher on settlement plates with deep/tall grooves and ridges or ledges (15 species/plate) than plates with shallow/short ones (10/plate) and plates without (8/plate), which were similar. At one site, the same was true for species diversity (data reported as Shannon index) and macroalgae and non-mobile invertebrate abundance (deep/tall: 77–99% cover; shallow/short: 30%; none: 31%). At the second site, no significant differences were found (deep/tall: 116–120%; shallow/short: 102%; none: 87%). Oyster (Ostreidae) abundance was higher on plates with grooves and ridges/ledges (52–91 individuals/plate) than without (15/plate), while mobile invertebrate abundance did not significantly differ (23–49 vs 11/plate). Twenty-three species (4 macroalgae, 14 mobile invertebrates, 5 non-mobile invertebrates) recorded on plates with grooves and ridges/ledges were absent from those without. The orientation of grooves and ridges or ledges had no clear effect on results. See paper for full results. Concrete settlement plates (250 × 250 mm) were moulded with and without groove habitats and small ridges or ledges. Plates with grooves and ridges or ledges had four grooves (length: 250 mm; width: 15–50 mm) between five vertical ridges or horizontal ledges (length: 250 mm; width: 17–65 mm). Grooves, ridges and ledges were either deep/tall (depth/height: 50 mm) or shallow/short (25 mm). Five of each were randomly arranged at midshore on each of two vertical sandstone seawalls in November 2015. Plates had textured surfaces. Macroalgae and invertebrates on plates were counted in the laboratory after 12 months.

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2.16. Create short flexible habitats (1–50 mm) on intertidal artificial structures

- **One study** examined the effects of creating short flexible habitats on intertidal artificial structures on the biodiversity of those structures. The study was in an estuary in southeast Australia[1].

COMMUNITY RESPONSE (1 STUDY)

- **Overall community composition (1 study):** One replicated, randomized, controlled study in Australia[1] found that creating short flexible habitats on intertidal artificial structures altered the combined macroalgae and non-mobile invertebrate community composition on structure surfaces, and had mixed effects on the combined mobile invertebrate and fish community composition on and around structure surfaces during low tide, depending on the site.

- **Invertebrate community composition (1 study):** One replicated, randomized, controlled study in Australia[1] found that creating short flexible habitats on intertidal artificial structures did not alter the mobile invertebrate community composition on and around structure surfaces during high tide.
- **Fish community composition (1 study):** One replicated, randomized, controlled study in Australia[1] found that creating short flexible habitats on intertidal artificial structures did not alter the fish community composition on and around structure surfaces during high tide.
- **Overall richness/diversity (1 study):** One replicated, randomized, controlled study in Australia[1] found that creating short flexible habitats on intertidal artificial structures decreased the combined macroalgae, invertebrate and fish species richness on and around structure surfaces during low tide.
- **Invertebrate richness/diversity (1 study):** One replicated, randomized, controlled study in Australia[1] found that creating short flexible habitats on intertidal artificial structures had mixed effects on the mobile invertebrate species richness on and around structure surfaces during high tide, depending on the site.
- **Fish richness/diversity (1 study):** One replicated, randomized, controlled study in Australia[1] found that creating short flexible habitats on intertidal artificial structures did not increase the fish species richness on and around structure surfaces during high tide.

POPULATION RESPONSE (1 STUDY)

- **Overall abundance (1 study):** One replicated, randomized, controlled study in Australia[1] found that creating short flexible habitats on intertidal artificial structures did not increase the combined mobile invertebrate and fish abundance on and around structure surfaces during low tide.
- **Algal abundance (1 study):** One replicated, randomized, controlled study in Australia[1] found that creating short flexible habitats on intertidal artificial structures had mixed effects on the macroalgal abundance on structure surfaces, depending on the species group and site.
- **Invertebrate abundance (1 study):** One replicated, randomized, controlled study in Australia[1] found that creating short flexible habitats on intertidal artificial structures had mixed effects on the abundance of non-mobile invertebrates on structure surfaces, and of mobile invertebrates during high tide, depending on the species group and site.
- **Fish abundance (1 study):** One replicated, randomized, controlled study in Australia[1] found that creating short flexible habitats on intertidal artificial structures did not increase the fish abundance on and around structure surfaces during high tide.

BEHAVIOUR (1 STUDY)

- **Fish behaviour change (1 study):** One replicated, randomized, controlled study in Australia[1] found that creating short flexible habitats on intertidal artificial structures did not increase the number of bites fishes took of structure surfaces.

Background

Short flexible habitats, such as understory macroalgal blades, turfs and soft-bodied invertebrates, provide other organisms refuge from desiccation and temperature fluctuations during low tide in intertidal rocky habitats (Kim 2002). They can support high biodiversity (Thrush *et al.* 2011) but can also dominate space and have negative effects on other species (O'Brien & Scheibling 2018). The size, density and material properties of flexible habitats are likely to affect the size, abundance and variety of organisms that can use them and the spaces they create.

Some organisms that form flexible habitats tend to be absent or sparse on intertidal artificial structures (Firth *et al.* 2016), although some readily colonize in suitable conditions. Artificial flexible habitats such as ropes or nets can be present on some structures, but are likely to be temporary and regularly disturbed (e.g. moved and replaced) when present. Short flexible habitats can be created on intertidal artificial structures by adding material, either during construction or retrospectively. In addition to potential biodiversity benefits, flexible habitats may offer some bioprotection for the underlying substrate, with potential to reduce weathering and enhance the durability of engineering materials (Coombes *et al.* 2013). Material choice is important for creating flexible habitats, since some flexible materials are unlikely to persist in the marine environment, while those that do may become entanglement hazards or contribute to pollution if dislodged. Studies that investigate the effects of transplanting live soft-bodied organisms onto structures are not included here, but are considered under the action "*Transplant or seed organisms onto intertidal artificial structures*".

There is a body of literature describing the use of artificial turfs as collectors to measure larval supply and settlement in rocky intertidal habitats and to investigate the effects of structural complexity on ecological interactions (e.g. Kelaher 2003; Morris *et al.* 2018; von der Meden *et al.* 2015). These studies are not included in this synopsis, which focusses on *in situ* conservation actions to enhance the biodiversity of marine artificial structures.

Definition: 'Short flexible habitats' are flexible protruding materials such as rope, ribbon or twine 1–50 mm in length (modified from "Soft structures" in Strain *et al.* 2018).

See also: *Create long flexible habitats (>50 mm) on intertidal artificial structures*; *Transplant or seed organisms onto intertidal artificial structures*.

Coombes M.A., Naylor L.A., Viles H.A. & Thompson R.C. (2013) Bioprotection and disturbance: seaweed, microclimatic stability and conditions for mechanical weathering in the intertidal zone. *Geomorphology*, 202, 4–14.

Firth L.B., White F.J., Schofield M., Hanley M.E., Burrows M.T., Thompson R.C., Skov M.W., Evans A.J., Moore P.J. & Hawkins S.J. (2016) Facing the future: the importance of substratum features for ecological engineering of artificial habitats in the rocky intertidal. *Marine and Freshwater Research*, 67, 131–143.

Kelaher B.P. (2003) Changes in habitat complexity negatively affect diverse gastropod assemblages in coralline algal turf. *Oecologia*, 135, 431–441.

Kim J.H. (2002) Patterns of interactions among neighbour species in a high intertidal algal community. *Algae*, 17, 41–51.

Morris R.L., Martinez A.S., Firth L.B. & Coleman R.A. (2018) Can transplanting enhance mobile marine invertebrates in ecologically engineered rock pools? *Marine Environmental Research*, 141, 119–127.

O'Brien J.M. & Scheibling R.E. (2018) Turf wars: competition between foundation and turf-forming species on temperate and tropical reefs and its role in regime shifts. *Marine Ecology Progress Series*, 590, 1–17.

Strain E.M.A., Olabarria C., Mayer-Pinto M., Cumbo V., Morris R.L., Bugnot A.B., Dafforn K.A., Heery E., Firth L.B., Brooks P.R. & Bishop M.J. (2018) Eco-engineering urban infrastructure for marine and coastal biodiversity: which interventions have the greatest ecological benefit? *Journal of Applied Ecology*, 55, 426–441.

Thrush S.F., Chiantore M., Asnaghi V., Hewitt J., Fiorentino D. & Cattaneo-Vietti R. (2011) Habitat-diversity relationships in rocky shore algal turf infaunal communities. *Marine Ecology Progress Series*, 424, 119–132.

von der Meden C.E.O., Cole V.J. & McQuaid C.D. (2015) Do the threats of predation and competition alter larval behaviour and selectivity at settlement under field conditions? *Journal of Experimental Marine Biology and Ecology*, 471, 240–246.

A replicated, randomized, controlled study in 2016 on two intertidal seawalls in Sydney Harbour estuary, Australia (1) found that adding short flexible habitats (coir panels) to rock pools created on the seawalls had mixed effects on macroalgae, invertebrate and fish community composition, species richness and abundances, depending on the species group and site. Over eight months, during low tide, a total of 44 macroalgae, invertebrate and fish species groups were recorded in pools with coir and 57 in pools without (data not statistically tested). Average macroalgae and non-mobile invertebrate species richness was lower in pools with coir (9 species/pool) than without (12/pool) and the community composition differed (data reported as statistical model results), while abundances varied depending on the species group and site (data not reported). Mobile invertebrate and fish species richness was also lower in pools with coir (2 species/pool) than without (3/pool), but their abundance was similar (data not reported), while effects on their community composition varied by site. During high tide, a total of 13 fish species were recorded in and around pools with coir and 14 in and around pools without, while 49 mobile invertebrate species groups were recorded in each. Average fish species richness, abundance, community composition, and the number of bites they took, were all similar in and around pools with and without coir (data not reported). Mobile invertebrate species richness in pools with coir (8–11 species/pool) and without (9–16/pool) varied by site, as did their abundances (data not reported), but the community composition was similar. Short flexible habitats (coir panels: 734 cm^2, 15 mm fibre length, 168 fibres/cm^2) were created on the inside vertical surfaces of concrete rock pools created on two vertical sandstone seawalls in January–February 2016. Five pools with coir and five without were randomly arranged at midshore in each of two sites along each seawall. Macroalgae, invertebrates and fishes were counted in pools during low tide over eight months. Mobile invertebrates and fishes were also surveyed during two high tides using a suction pump and videos, respectively. Three pools were missing and no longer provided habitat.

(1) Morris R.L., Golding S., Dafforn K.A. & Coleman R.A. (2018) Can coir increase native biodiversity and reduce colonisation of non-indigenous species in eco-engineered rock pools? *Ecological Engineering*, 120, 622–630.

2.17. Create long flexible habitats (>50 mm) on intertidal artificial structures

- **One study** examined the effects of creating long flexible habitats on intertidal artificial structures on the biodiversity of those structures. The study was in a port in the Netherlands[1].

COMMUNITY RESPONSE (1 STUDY)

- **Overall community composition (1 study):** One replicated, controlled study in the Netherlands[1] reported that creating long flexible habitats on intertidal artificial structures altered the combined macroalgae and non-mobile invertebrate community composition on structure surfaces. The flexible habitats themselves supported macroalgae, mobile and non-mobile invertebrates that were absent from structure surfaces without flexible habitats.

POPULATION RESPONSE (0 STUDIES)

BEHAVIOUR (0 STUDIES)

Background

Long flexible habitats, such as macroalgal canopies and soft-bodied invertebrates, provide other organisms refuge from desiccation and temperature fluctuations during low tide in intertidal rocky habitats (Moore *et al.* 2007). They also provide shelter from predation (Dumas & Witman 1993). The size, density and material properties of flexible habitats are likely to affect the size, abundance and variety of organisms that can use them and the spaces they create.

Some organisms that form flexible habitats tend to be absent or sparse on intertidal artificial structures (Firth *et al.* 2016), although some readily colonize in wave-sheltered conditions (Jonsson *et al.* 2006). Artificial flexible habitats such as ropes or nets can be present on some structures, but are likely to be temporary and regularly disturbed (e.g. removed and replaced) when present. Long flexible habitats can be created on intertidal artificial structures by adding material, either during construction or retrospectively. In addition to potential biodiversity benefits, flexible habitats may offer some bioprotection for the underlying substrate, with potential to reduce weathering and enhance the durability of engineering materials (Coombes *et al.* 2013). Material choice is important for creating flexible habitats, since some flexible materials are unlikely to persist in the marine environment, while those that do may become entanglement hazards or contribute to pollution if dislodged. Studies that investigate the effects of transplanting live soft-bodied organisms onto structures are not included here, but are considered under the action "*Transplant or seed organisms onto intertidal artificial structures*".

Definition: 'Long flexible habitats' are flexible protruding materials such as rope, ribbon or twine >50 mm in length (modified from "Soft structures" in Strain *et al.* 2018).

See also: *Create short flexible habitats (1–50 mm) on intertidal artificial structures*; *Transplant or seed organisms onto intertidal artificial structures*.

Coombes M.A., Naylor L.A., Viles H.A. & Thompson R.C. (2013) Bioprotection and disturbance: seaweed, microclimatic stability and conditions for mechanical weathering in the intertidal zone. *Geomorphology*, 202, 4–14.
Dumas J.V. & Witman J.D. (1993) Predation by herring gulls (*Larus argentatus* Coues) on two rocky intertidal crab species [*Carcinus maenas* (L.) & *Cancer irroratus* Say]. *Journal of Experimental Marine Biology and Ecology*, 169, 89–101.
Firth L.B., White F.J., Schofield M., Hanley M.E., Burrows M.T., Thompson R.C., Skov M.W., Evans A.J., Moore P.J. & Hawkins S.J. (2016) Facing the future: the importance of substratum features for ecological engineering of artificial habitats in the rocky intertidal. *Marine and Freshwater Research*, 67, 131–143.
Jonsson P.R., Granhag L., Moschella P.S., Åberg P., Hawkins S.J. & Thompson R.C. (2006) Interactions between wave action and grazing control the distribution of intertidal macroalgae. *Ecology*, 87, 1169–1178.
Moore P., Hawkins S.J. & Thompson R.C. (2007) Role of biological habitat amelioration in altering the relative responses of congeneric species to climate change. *Marine Ecology Progress Series*, 334, 11–19.
Strain E.M.A., Olabarria C., Mayer-Pinto M., Cumbo V., Morris R.L., Bugnot A.B., Dafforn K.A., Heery E., Firth L.B., Brooks P.R. & Bishop M.J. (2018) Eco-engineering urban infrastructure for marine and coastal biodiversity: which interventions have the greatest ecological benefit? *Journal of Applied Ecology*, 55, 426–441.

One replicated, controlled study in 2009 on five intertidal jetty pilings in the Port of Rotterdam, the Netherlands (1) reported that creating long flexible habitats ('hulas') on pilings altered the macroalgae and non-mobile invertebrate community composition on piling surfaces, and that hulas were colonized by macroalgae and invertebrates, but data were not statistically tested. After eight months, hula ropes supported mussels (*Mytilus edulis*: 5% cover), barnacles (*Amphibalanus improvisus*: 1%), red macroalgae (*Ceramium rubrum*: 0.2%) and amphipods (Amphipoda: 11–100 individuals/rope), which were all absent from piling surfaces without flexible habitats. Average biomass on ropes was 1 g/cm. Piling surfaces under hulas supported mostly barnacles (50% cover), while pilings without flexible habitats supported mostly green macroalgae (50% cover). Long flexible habitats were created by attaching polyamide rope skirts ('hulas') around pilings in March 2009. One hula with 167 ropes (diameter: 6 mm; length: 550 mm; density: 167/m) was attached at lowshore around each of five wooden pilings, cleared of organisms. Hulas were compared with intertidal surfaces (200 × 200 mm) on five pilings without flexible habitats, cleared of organisms. Macroalgae and invertebrates on hula ropes and piling surfaces were counted and biomass (wet weight) measured in the laboratory over eight months.

(1) Paalvast P., van Wesenbeeck B.K., van der Velde G. & de Vries M.B. (2012) Pole and pontoon hulas: an effective way of ecological engineering to increase productivity and biodiversity in the hard-substrate environment of the port of Rotterdam. *Ecological Engineering*, 44, 199–209.

2.18. Reduce the slope of intertidal artificial structures

- **Two studies** examined the effects of reducing the slope of intertidal artificial structures on the biodiversity of those structures. The studies were in an estuary in southeast Australia[1,2].

COMMUNITY RESPONSE (1 STUDY)

- **Overall richness/diversity (1 study):** One before-and-after study in Australia[2] reported that reducing the slope of an intertidal artificial structure, along with creating rock pools, increased the combined macroalgae, invertebrate and fish species richness on the structure.

POPULATION RESPONSE (1 STUDY)

- **Algal abundance (1 study):** One replicated, controlled study in Australia[1] found that reducing the slope of an intertidal artificial structure did not increase the macroalgal abundance on structure surfaces.
- **Invertebrate abundance (1 study):** One replicated, controlled study in Australia[1] found that reducing the slope of an intertidal artificial structure did not increase the oyster or mobile invertebrate abundance on structure surfaces.

BEHAVIOUR (0 STUDIES)

Background

The slope of substrate surfaces can influence the species that colonize in intertidal rocky habitats (Harmelin-Vivien *et al*. 1995; Vaselli *et al.* 2008; but see Cacabelos *et al.* 2016; Firth *et al.* 2016). Artificial structures tend to have steeper slopes than natural reefs, with narrower bands of intertidal habitat. This means that space for organisms is scarce and competitive interactions and other environmental processes differ (Chapman & Underwood 2011). Steep surfaces can also be associated with the presence of non-native species (Dafforn 2017).

Although fundamental aspects of structure designs, such as their slope, are likely to be driven by engineering and cost requirements, there may be opportunities to reduce the slope of intertidal artificial structure surfaces with the aim of enhancing their biodiversity. This may, however, lead to an increase in the physical footprint of structures and associated impacts on the receiving environment (Perkins *et al.* 2015). For this reason, studies that test the effects of creating additional artificial habitat in front of existing structures to create horizontal or gently sloping surfaces are not included in this synopsis, although such actions can deliver biodiversity benefits and are informative (e.g. Liversage & Chapman 2018; Toft *et al.* 2013).

Definition: 'Reducing the slope' includes actions taken to reduce the inclination of structures without increasing the footprint, with the aim of enhancing their biodiversity.

See also: *Create small protrusions (1–50 mm) on intertidal artificial structures*; *Create large protrusions (>50 mm) on intertidal artificial structures*; *Create small ridges or ledges*

(1–50 mm) on intertidal artificial structures; *Create large ridges or ledges (>50 mm) on intertidal artificial structures*.

Cacabelos E., Martins G.M., Thompson R., Prestes A.C.L., Azevedo J.M.N. & Neto A.I. (2016) Factors limiting the establishment of canopy-forming algae on artificial structures. *Estuarine, Coastal and Shelf Science*, 181, 277–283.

Chapman M.G. & Underwood A.J. (2011) Evaluation of ecological engineering of "armoured" shorelines to improve their value as habitat. *Journal of Experimental Marine Biology and Ecology*, 400, 302–313.

Dafforn K.A. (2017) Eco-engineering and management strategies for marine infrastructures to reduce establishment and dispersal of non-indigenous species. *Management of Biological Invasions*, 8, 153–161.

Firth L.B., White F.J., Schofield M., Hanley M.E., Burrows M.T., Thompson R.C., Skov M.W., Evans A.J., Moore P.J. & Hawkins S.J. (2016) Facing the future: the importance of substratum features for ecological engineering of artificial habitats in the rocky intertidal. *Marine and Freshwater Research*, 67, 131–143.

Harmelin-Vivien M.L., Harmelin J.G. & Leboulleux V. (1995) Microhabitat requirements for settlement of juvenile sparid fishes on Mediterranean rocky shores. *Hydrobiologia*, 300, 309–320.

Liversage K. & Chapman M.G. (2018) Coastal ecological engineering and habitat restoration: incorporating biologically diverse boulder habitat. *Marine Ecology Progress Series*, 593, 173–185.

Perkins M.J., Ng T.P.T., Dudgeon D., Bonebrake T.C. & Leung K.M.Y. (2015) Conserving intertidal habitats: what is the potential of ecological engineering to mitigate impacts of coastal structures? *Estuarine, Coastal and Shelf Science*, 167, 504–515.

Toft J.D., Ogston A.S., Heerhartz S.M., Cordell J.R. & Flemer E.E. (2013) Ecological response and physical stability of habitat enhancements along an urban armored shoreline. *Ecological Engineering*, 57, 97–108.

Vaselli S., Bertocci I., Maggi E. & Benedetti-Cecchi L. (2008) Assessing the consequences of sea level rise: effects of changes in the slope of the substratum on sessile assemblages of rocky seashores. *Marine Ecology Progress Series*, 368, 9–22.

A replicated, controlled study (year not reported) on an intertidal seawall in Sydney Harbour estuary, Australia (1) found that reducing the slope of the seawall did not increase the abundance of macroalgae, oysters *Saccostrea glomerata* or mobile invertebrates on seawall surfaces. Over 24 months, the abundances of macroalgae, oysters and mobile invertebrates were similar on surfaces of a new sloping seawall and on remnants of the original vertical wall that it replaced (data not reported). The slope of a seawall was reduced by replacing a vertical concrete wall with a sloping wall of boulders. This increased the extent of the intertidal area from high to low shore by 2–3 m (timing and other details of the intervention not reported). Macroalgae and invertebrates were counted on 10 surfaces (dimensions not reported) in each of four sites on the new sloping wall, and 10 on a remnant of the original vertical wall, during low tide over 24 months.

A before-and-after study in 2012–2013 on an intertidal seawall in Sydney Harbour estuary, Australia (2) reported that reducing the slope of the seawall, along with creating rock pools on the wall, increased the macroalgae, invertebrate and fish species richness on the wall. A total of 25 macroalgae, invertebrate and fish species were recorded on the seawall and in pools after the slope was reduced and pools were created, compared with 10 species on the seawall before (data not statistically tested). It is not clear whether these effects were the direct result of reducing the slope of the seawall or creating rock pools. The slope of a sandstone boulder seawall was reduced during reconstruction in July 2012 (details not reported). Three large rock pools (area: 2 m^2; depth: 300 mm;

volume: 600 l) were also created on the wall. Macroalgae, invertebrates and fishes were counted during low tide on the wall before reconstruction and on the wall and in pools after reconstruction in 2013 (sampling details and month not reported).

(1) Chapman M.G. & Underwood A.J. (2011) Evaluation of ecological engineering of "armoured" shorelines to improve their value as habitat. *Journal of Experimental Marine Biology and Ecology*, 400, 302–313.
(2) Heath T. & Moody G. (2013) *Habitat development along a highly urbanised foreshore.* Proceedings of the New South Wales Coastal Conference 2013, Sydney, Australia, 1–7.

2.19. Transplant or seed organisms onto intertidal artificial structures

- **Ten studies** examined the effects of transplanting or seeding species onto intertidal artificial structures on the biodiversity of those structures. Seven studies were in estuaries in southeast Australia[2,4,5,6,9,10] and Hong Kong[7], two were on island coastlines in the Singapore Strait[3,8] and one was in a port and on an open coastline in southeast Spain[1].

COMMUNITY RESPONSE (5 STUDIES)

- **Overall community composition (3 studies):** Three replicated, randomized, controlled studies in Hong Kong[7] and Australia[9,10] reported that oysters transplanted onto intertidal artificial structures supported macroalgae[9,10], mobile invertebrate[7,9,10], non-mobile invertebrate[9,10] and fish[9] species that were absent from on and around structure surfaces without transplanted oysters.
- **Overall richness/diversity (3 studies):** Three replicated, randomized, controlled studies in Hong Kong[7] and Australia[9,10] found that transplanting oysters onto intertidal artificial structures had mixed effects on the combined macroalgae and invertebrate species richness[7,9,10] and/or diversity[10] on structure surfaces, depending on the site[7] and/or the presence[7,9,10] and size[10] of grooves and small ridges[7,10] or ledges[9,10] on surfaces.
- **Invertebrate richness/diversity (1 study):** One replicated, randomized, controlled study in Australia[9] found that transplanting oysters onto intertidal artificial structures increased the mobile invertebrate species richness on structure surfaces.
- **Fish richness/diversity (3 studies):** Two of three replicated, randomized studies (including two controlled studies) in Australia[5,6,9] found that transplanting oysters[6,9] and/or coralline algae[6] onto intertidal artificial structures did not increase the fish species richness on and around structure surfaces. One[5] found mixed effects of transplanting oysters, depending on the presence and size of grooves and small ridges on surfaces and the site.

POPULATION RESPONSE (10 STUDIES)

- **Overall abundance (2 studies):** One of two replicated, randomized, controlled studies in Australia[9,10] found that transplanting oysters onto intertidal artificial structures did not increase the combined macroalgae and invertebrate abundance on structure surfaces[9]. One study[10] found mixed effects depending on the presence and size of grooves and small ridges/ledges on structure surfaces.

- **Invertebrate abundance (3 studies):** Two of three replicated, randomized, controlled studies in Hong Kong[7] and Australia[9,10] found that transplanting oysters onto intertidal artificial structures had mixed effects on the mobile invertebrate abundance on structure surfaces, depending on the presence of grooves and small ridges[7] or ledges[9] on surfaces and/or the site[7]. One of the studies[7] also found that transplanting oysters increased the non-mobile invertebrate and oyster recruit abundance and decreased barnacle abundance. One[10] found increased oyster and mobile invertebrate abundance.
- **Fish abundance (3 studies):** Two of three replicated, randomized studies (including two controlled studies) in Australia[5,6,9] found that transplanting oysters[6,9] and/or coralline algae[6] onto intertidal artificial structures did not increase the fish abundance on and around structure surfaces. One[5] found that fish abundance around transplanted oysters was similar regardless of the presence and size of grooves and small ridges on structure surfaces.
- **Algal survival (1 study):** One replicated study in Singapore[8] found that macroalgae transplanted onto an intertidal artificial structure were more likely to survive at mid- and highshore than at lowshore.
- **Invertebrate survival (8 studies):** Six of eight studies (including six replicated, three randomized and two controlled studies) in Australia[2,4,5,9,10], Spain[1], Singapore[3] and Hong Kong[7] reported that the survival of mobile invertebrates (seasnails[2,4], starfish[2,4] and/or urchins and anemones[2]) or non-mobile invertebrates (limpets[1], corals and sponges[3] or oysters[9]) transplanted onto intertidal artificial structures varied depending on the species[2,3,4], site[1,5], and/or the presence[5,9] and size[5] of grooves and small ridges[5] or ledges[9] on structure surfaces. One of the studies[5] found that oyster survival was higher when transplanted into grooves compared with on ridges, while one[9] found that survival in grooves and on ledges varied depending on the site. Two studies[7,10] simply reported that a proportion of transplanted oysters survived.
- **Algal condition (1 study):** One replicated study in Singapore[8] found that the growth of macroalgae transplanted onto an intertidal artificial structure was similar at lowshore, midshore and highshore.
- **Invertebrate condition (2 studies):** One study in Singapore[3] reported that the growth of corals and sponges transplanted onto an intertidal artificial structure varied depending on the species. One replicated study in Spain[1] simply reported that transplanted limpets grew.

BEHAVIOUR (1 STUDY)

- **Fish behaviour change (1 study):** One replicated, randomized, controlled study in Australia[6] found that transplanting oysters and/or coralline algae onto intertidal artificial structures did not increase the time fishes spent interacting with structure surfaces or the number of bites they took, but that benthic fishes took more bites from surfaces with transplanted oysters than from those with transplanted algae and oysters together. These results were true regardless of whether there were grooves and small ridges on structure surfaces.

Background

Natural intertidal rocky habitats tend to support many more species than do artificial structures (Moschella *et al.* 2005). Species can be transplanted or seeded directly onto structures with the aim of enhancing their biodiversity. The choice of species to transplant or seed depends on overarching management objectives. There may be value in transplanting rare, threatened, or commercially-valuable species to create artificial surrogate habitats for localized populations. Alternatively, keystone species such as primary producers, grazers, habitat-providers and water-quality regulators may be of interest to create productive and well-regulated artificial ecosystems. Species common to natural reefs may be the focus if, for example, larvae/spores cannot disperse far enough or hydrographic barriers (currents/tides) prevent recruitment, or to pre-empt potential non-native species invasions on new substrates.

When planning transplanting or seeding interventions, the method and timing of intervention and the suitability of the receiving environment should be carefully considered, as these factors may impact the likelihood of success. It is crucial to understand the reasons for a species' absence in the first place. If, for example, a species has declined due to poor water quality and there has been no intervening improvement, there can be little expectation of successful recovery following transplantation or seeding (Stevenson *et al.* 1993). Similarly, species that naturally occur in wave-sheltered habitats should not be expected to survive and thrive on wave-exposed structures (Jonsson *et al.* 2006). Biological factors such as predation pressure, competition and food availability may also affect outcomes and should be understood prior to carrying out the intervention (Gianni *et al.* 2018).

There are bodies of literature describing transplanting or seeding species into natural or artificial intertidal habitats to investigate ecological processes and interactions (e.g. Dudgeon & Petraitis 2005; Iveša *et al.* 2010; Wilkie *et al.* 2012), for restoration in natural habitats (e.g. Vanderklift *et al.* 2020) and for artificial reefs (e.g. Walles *et al.* 2016). These studies are not included in this synopsis, which focusses on *in situ* conservation actions to enhance the biodiversity of structures that are engineered to fulfil a primary function other than providing artificial habitats.

Definition: 'Transplanting or seeding species' includes actions taken to attach live organisms at any life stage onto structures, with the aim of generating self-sustaining populations.

Dudgeon S. & Petraitis P.S. (2005) First year demography of the foundation species, *Ascophyllum nodosum*, and its community implications. *Oikos*, 109, 405–415.

Gianni F., Bartolini F., Airoldi L. & Mangialajo L. (2018) Reduction of herbivorous fish pressure can facilitate focal algal species forestation on artificial structures, *Marine Environmental Research*, 138, 102–109.

Iveša L., Chapman M.G., Underwood A.J. & Murphy R.J. (2010) Differential patterns of distribution of limpets on intertidal seawalls: experimental investigation of the roles of recruitment, survival and competition. *Marine Ecology Progress Series*, 407, 55–69.

Jonsson P.R., Granhag L., Moschella P.S., Åberg P., Hawkins S.J. & Thompson R.C. (2006) Interactions between wave action and grazing control the distribution of intertidal macroalgae. *Ecology*, 87, 1169–1178.

Moschella P.S., Abbiati M., Åberg P., Airoldi L., Anderson J.M., Bacchiocchi F., Bulleri F., Dinesen G.E., Frost M., Gacia E., Granhag L., Jonsson P.R., Satta M.P., Sundelöf A., Thompson R.C. & Hawkins S.J. (2005) Low-crested coastal defence structures as artificial habitats for marine life: using ecological criteria in design. *Coastal Engineering*, 52, 1053–1071.

Stevenson J.C., Staver L.W. & Staver K.W. (1993) Water quality associated with survival of submersed aquatic vegetation along an estuarine gradient. *Estuaries*, 16, 346–361.

Venderklift M.A., Doropoulos C., Gorman D., Leal I., Minne A.J.P., Statton J., Steven A.D.L. & Wernberg T. (2020) Using propagules to restore coastal marine ecosystems. *Frontiers in Marine Science*, 7, 724.

Walles B., Troost K., van den Ende D., Nieuwhof S., Smaal A.C. & Ysebaert T. (2016) From artificial structures to self-sustaining oyster reefs. *Journal of Sea Research*, 108, 1–9.

Wilkie E.M., Bishop M.J. & O'Connor W.A. (2012) Are native *Saccostrea glomerata* and invasive *Crassostrea gigas* oysters habitat equivalents for epibenthic communities in south-eastern Australia? *Journal of Experimental Marine Biology and Ecology*, 420–421, 16–25.

A replicated study in 2003–2005 on four intertidal breakwaters in Ceuta Port and on open coastline in the Alboran Sea, Spain (1) reported that 0–17% of limpets *Patella ferruginea* transplanted onto breakwaters survived, but that most survivors grew. Data were not statistically tested. After 28 months, 2–17% of transplanted limpets survived on the wave-sheltered inaccessible breakwater, 15% on the wave-exposed inaccessible breakwater, 8% on the wave-sheltered accessible breakwater, and 0% on the wave-exposed accessible breakwater. Growth rates ranged from 0–4 mm/month with no clear differences between sites. Limpets were collected from a boulder breakwater during reconstruction, marked and then transplanted onto four nearby boulder breakwaters. Twenty limpets were transplanted onto boulder surfaces in each of three patches (10 m long) on each of four breakwaters during spring 2003 (shore level/month and other transplantation details not reported). Breakwaters were either inside the port (wave-sheltered) or outside (wave-exposed) and either accessible to people or inaccessible, with one breakwater/exposure-accessibility combination. Transplants were monitored over 28 months.

A study in 2010 on two intertidal seawalls in Sydney Harbour estuary, Australia (2) reported that the survival of mobile invertebrates transplanted into rock pools created on the seawalls varied depending on the species. After 10 days, transplanted turban snails *Turbo undulatus*, keyhole limpets *Scutus antipodes* and sea anemones *Actinia tenebrosa* survived in midshore pools on both seawalls (data not reported). All transplanted sea urchins *Heliocidaris erythrogramma* and starfish *Patiriella calcar* had died and no transplanted nerite snails *Nerita atramentosa* remained in pools. Six species of mobile invertebrates were collected from natural reefs and transplanted into rock pools created on two vertical sandstone seawalls at highshore and midshore in 2010. No other details were reported. Transplanted animals were surveyed during low tide after 10 days.

A study in 2010–2012 on an intertidal seawall on an island coastline in the Singapore Strait, Singapore (3) found that 0–90% of coral and sponge fragments transplanted onto the seawall survived, depending on the species, and that most survivors grew. After 24 months, hard coral transplant survival was higher for *Porites lobata* (47%) than *Pocillopora damicornis* and *Hydnophora rigida* (both 0%). Soft coral survival was higher for *Lobophytum* sp. (88%) than *Cladiella* sp. (37%) and *Sinularia* sp. (13%). Sponge survival was higher for *Lendenfeldia chondrodes* (68%) than *Spongia ceylonensis* (14%) and *Rhabdastrella globostellata* (0%). In hard corals transplanted for 13 months, survival was higher for *Goniastrea minuta* (90%) than *Diploastrea heliopora* (10%). *Diploastrea heliopora* fragments had negative growth rates (-1.2 cm^2/month), while the other seven surviving species had positive growth rates (0.3–19.7 cm^2/month). Corals and sponges were collected from natural reefs, reared in a nursery, then fragmented and transplanted onto a granite boulder seawall at lowshore. Fragments (≥30 mm) of three hard coral species (18–38 fragments/species), three soft coral species (30–40/species) and three sponge species (44–49/species) were transplanted in May 2010. Fragments of two additional hard coral species (30 fragments/species) were transplanted in April 2011. Hard corals were transplanted directly onto seawall surfaces using epoxy putty, while soft corals and sponges were grown onto concrete plates (50 mm diameter, 5 mm thick) first. Transplants were monitored during low tide until May 2012.

A replicated study in 2016 on an intertidal seawall in Sydney Harbour estuary, Australia (4) reported that 18–79% of mobile invertebrates transplanted into rock pools created on the seawall remained in and around the pools, depending on the species. After one day, on average, 60% of transplanted topshells *Austrocochlea porcata* remained in pools (30%) and on seawall surfaces around pools (30%). Between 73–79% of transplanted periwinkles *Bembicium nanum* remained in (23–29%) and around (50%) pools. Only 18% of transplanted starfish *Parvulastra exigua* remained in pools and none around pools. Topshells, periwinkles and starfish were collected from natural rock pools and transplanted into artificial pools created at midshore on a vertical sandstone seawall. Ten individuals of each species were transplanted into each of three pools on each of two occasions during January–February 2016. Transplanted animals remaining in and around pools on the seawall were counted during low tide after one day.

A replicated, randomized study in 2015–2016 on two intertidal seawalls in Sydney Harbour estuary, Australia (5) found that 28–94% of oysters *Saccostrea glomerata* transplanted onto settlement plates survived, and that oyster survival and fish species richness around plates, but not fish abundance, varied depending on the presence and size of grooves and small ridges on plates and the site. Over six months, at one of two sites, transplanted oyster survival was higher on settlement plates with deep/tall grooves and ridges (52%) than on plates without (28%), and both were similar to plates with shallow/short grooves and ridges (43%). Survival was higher in grooves (85–95%) than on ridges (30–35%). Fish species richness was higher on and around oyster plates with deep/tall grooves and ridges (7 species/plate) than without (4/plate), while maximum fish abundance was similar (5 vs 4 individuals/plate) (not reported for shallow/short grooves and ridges). At the second site, no significant differences were found for oyster survival (deep/tall grooves and ridges: 94%; shallow/short: 80%; none: 91%; grooves: 96–98%; ridges: 95–98%), fish species richness (3 species/plate with and without grooves and ridges) or fish abundance (2 vs 3 individuals/plate). Dead oysters were either cracked (0–60%), intact (0–5%) or missing (0–8%). Hatchery-reared juvenile oysters were attached to concrete settlement plates (250 × 250 mm) using epoxy glue and transplanted onto vertical sandstone seawalls. There were 52 oysters/plate (24 mm average length) in patches of 4–5 individuals covering ~220 cm^2. Plates had textured surfaces with or without deep/tall (50 mm) or shallow/short (25 mm) grooves and small ridges. Five of each were randomly arranged at midshore on each of two seawalls in November 2015. Transplanted oysters were monitored during low tide over six months. Fishes were counted on and around plates from time-lapsed photographs during two high tides after one month.

A replicated, randomized, controlled study in 2016–2017 on three intertidal seawalls in Sydney Harbour estuary, Australia (6) found that transplanting oysters *Saccostrea glomerata*, coralline algae *Corallina officinalis*, or both onto settlement plates did not increase the fish species richness or abundance or alter fish behaviour on and around plates, but benthic fish behaviour varied depending on the species transplanted. After 8–12 months, fish species richness and abundance were similar on and around settlement plates with and without transplanted coralline algae, oysters or both (data not reported). The same was true for the time fishes spent interacting with plates (with coralline algae: 1–21 minutes/60-minute survey; with oysters: 2–30 minutes/survey; both: 1–18/survey; neither: 1–27/survey). Benthic fishes took more bites from plates with oysters (10 bites/survey) than plates with both algae and oysters (2/survey), while both were similar to plates with algae only (6 bites/survey) and with neither (4/survey). There were no significant differences for pelagic fishes (with algae: 8 bites/survey; oysters: 21/survey; both: 5/survey; neither: 8/survey). Coralline algae collected from natural reefs and hatchery-reared juvenile oysters were attached to concrete settlement plates (250 × 250 mm) using epoxy glue and transplanted onto vertical sandstone seawalls. Algae, oysters (46 mm average length), both or neither were attached in eight patches/plate covering 125 cm^2. Plates had textured surfaces with or without grooves and small ledges (50 mm). Nine of each transplant-grooves/ledges combination were randomly arranged at midshore on each of three seawalls in March 2016. Fishes were counted on and around one of each plate design from 60-minute videos during each of three high tides after 8–12 months. The time fishes spent within 50 mm of plates and the number of bites they took was recorded.

A replicated, randomized, controlled study in 2016–2017 on two intertidal seawalls in the Pearl River estuary, Hong Kong (7) found that 36% of oysters *Saccostrea cuccullata* transplanted onto settlement plates survived, and that transplanting oysters increased non-mobile invertebrate and oyster abundance on plates, decreased barnacle abundance, and had mixed effects on mobile invertebrate abundance and macroalgae and invertebrate species richness, depending on the presence of grooves and small ridges on plates and the site. After 12 months, 36% of transplanted oysters survived. At one of two sites, settlement plates with transplanted oysters supported higher macroalgae and invertebrate species richness (14–17 species/plate) and mobile invertebrate abundance (51–106 individuals/plate) than plates without oysters (9–12 species/plate, 15–81 individuals/plate). At the second site, the same was true for flat plates (with oysters: 16 species/plate, 65 individuals/plate; without: 9 species/plate, 19 individuals/plate) but no significant differences were found for plates with grooves and ridges (12–13 species/plate with and without oysters, 43–49 vs 31–45 individuals/plate). At both sites, plates with transplanted oysters supported higher abundance of non-mobile invertebrates (48–58% cover) and new oyster recruits (1–4 g/plate) but fewer barnacles

(Cirripedia) (0–4 g/plate) than plates without (non-mobile invertebrates: 13–42%; oyster recruits: 0–2 g/plate; barnacles: 1–13 g/plate). Three mobile invertebrate species recorded on plates with transplanted oysters were absent from those without. Oysters collected locally were attached to concrete settlement plates (250 × 250 mm) using epoxy glue and transplanted onto vertical concrete seawalls. Plates had oyster patches covering 225 cm^2/plate or no oysters, and textured surfaces with or without deep/tall (50 mm) or shallow/short (25 mm) grooves and small ridges. Five of each transplant-grooves/ridges combination were randomly arranged at midshore on each of two seawalls in November 2016 (month/year: M. Perkins *pers. comms.*). Macroalgae and invertebrates on plates were counted from photographs and in the laboratory, and barnacle and oyster recruit biomass (dry weight) was measured after 12 months. One plate was missing and no longer provided habitat.

A replicated study in 2019 on an intertidal seawall on an island coastline in the Singapore Strait, Singapore (8) found that red macroalgae *Hydropuntia edulis* transplanted onto the seawall grew at similar rates at all shore levels, but was more likely to be dislodged at lowshore than at mid- and highshore. Over one month, the biomass of transplanted macroalgae increased by 2 g/individual on average. The average growth rate was 3%/day and average biomass yield was 2 kg/m^2 of seawall. Growth rates were similar at lowshore (3%/day), midshore (2%/day) and highshore (2%/day), but the probability of dislodgement was higher at lowshore (58%) than midshore (8%) and highshore (17%). Red macroalgae collected from natural reefs were woven into nylon ropes and transplanted into water-retaining plastic troughs (1.0 × 0.1 m) attached to a seawall. Six individuals were transplanted into each of four troughs at each of lowshore, midshore and highshore in January 2019. Growth rates (% change in wet weight/day) and biomass yield (change in wet weight/m^2) were measured during low tide after one month.

A replicated, randomized, controlled study in 2015–2016 on two intertidal seawalls in Sydney Harbour estuary, Australia (9) reported that 17–52% of oysters *Saccostrea glomerata* transplanted onto settlement plates survived, and found that oyster survival, macroalgae, invertebrate and fish species richness and abundances varied depending on the presence of grooves and small ledges on plates, the species group and site. Over 12 months, at one of two sites, transplanted oyster survival was higher on settlement plates with grooves and ledges (52%) than without (17%), and higher in grooves (92%) than on ledges (23%). At the second site, no significant differences were found (69% with and without grooves and ledges; grooves: 77%; ledges: 85%). Macroalgae and non-mobile invertebrate abundance was similar on plates with transplanted oysters (24–39% cover) and without (31–46%). Their species richness was higher on flat plates with oysters (5 species/plate) than those without (2/plate), but no significant difference was found when grooves and ledges were present on plates (6/plate with and without oysters). Mobile invertebrate species richness was higher on plates with oysters (10–12 species/plate) than without (4–7/plate). Their abundance was higher on plates with oysters (38 individuals/plate) than without (20/plate) when grooves and ledges were present, but did not differ on flat plates (16/plate with and without oysters). Fish species richness and abundance were similar on and around plates with and without oysters (3 vs 2 species/plate, both 1 individual/plate). Twenty-two species (3 macroalgae, 2 non-mobile invertebrates, 14 mobile invertebrates, 3 fishes) recorded on and around plates with transplanted oysters were absent from those without. Hatchery-reared juvenile oysters were attached to concrete settlement plates (250 × 250 mm) using epoxy glue and transplanted onto vertical sandstone seawalls. Plates had 52 oysters/plate in patches of 4–5 individuals or no oysters, and textured surfaces with or without grooves and small ledges (50 mm). Five of each transplant-grooves/ledges combination were randomly arranged at midshore on each of two seawalls in November 2015. Macroalgae and invertebrates were counted on plates during low tide, from photographs and in the laboratory over 12 months. Fishes were counted on and around plates from time-lapsed photographs during two high tides.

A replicated, randomized, controlled study in 2015–2016 on two intertidal seawalls in Sydney Harbour estuary, Australia (10) reported that 60% of oysters *Saccostrea glomerata* transplanted onto settlement plates survived, and found that macroalgae and invertebrate species richness, diversity and abundances varied depending on the presence and depth/height of grooves and ridges or ledges on plates, the species group and site. After 12 months, 60% of transplanted oysters survived. The macroalgae and invertebrate species richness was higher on settlement plates with transplanted oysters (18–19 species/plate) than without (8–10/plate) when there were no or shallow/short grooves and ridges or ledges on plates. When deep/tall grooves and ridges or ledges were present, richness was similar on plates with and without transplanted oysters (17 vs 15/plate). The same was true for species diversity (data reported as Shannon index) and at one site for macroalgae and non-mobile invertebrate abundance (no or short/shallow grooves and ridges: 102–128% cover with oysters vs 30% without; deep/tall grooves and ridges: 98–108% with oysters vs 77–99% without). At the second site, no significant differences were found (with oysters: 126–150%; without: 87–121%). Oyster (Ostreidae) and mobile invertebrate abundances were higher on plates with transplanted oysters (oysters: 101–152 individuals/plate; mobiles: 83–156/plate) than without (oysters: 15–91/plate; mobiles: 12–38/plate). Eighteen species (2 macroalgae, 15 mobile invertebrates, 1 non-mobile invertebrate) recorded on plates with transplanted oysters were absent from those without. See paper for full results. Juvenile oysters were attached to concrete settlement plates (250 × 250 mm) using epoxy glue and transplanted onto vertical sandstone seawalls. Plates had 52 oysters/plate in patches of 4–5 individuals or no oysters, and textured surfaces with or without deep/tall (50 mm) or shallow/short (25 mm) grooves and small ridges or ledges. Five of each transplant-grooves/ridges combination were randomly arranged at midshore on each of two seawalls in November 2015. Macroalgae and invertebrates on plates were counted in the laboratory after 12 months.

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(2) Browne M.A. & Chapman M.G. (2014) Mitigating against the loss of species by adding artificial intertidal pools to existing seawalls. *Marine Ecology Progress Series*, 497, 119–129.
(3) Ng C.S.L., Lim S.C., Ong J.Y., Teo L.M.S., Chou L.M., Chua K.E. & Tan K.S. (2015) Enhancing the biodiversity of coastal defence structures: transplantation of nursery-reared reef biota onto intertidal seawalls. *Ecological Engineering*, 82, 480–486.
(4) Morris R.L., Martinez A.S., Firth L.B. & Coleman R.A. (2018) Can transplanting enhance mobile marine invertebrates in ecologically engineered rock pools? *Marine Environmental Research*, 141, 119–127.
(5) Strain E.M.A., Morris R.L., Coleman R.A., Figueira W.F., Steinberg P.D., Johnston E.L. & Bishop M.J. (2018) Increasing microhabitat complexity on seawalls can reduce fish predation on native oysters. *Ecological Engineering*, 120, 637–644.
(6) Ushiama S., Mayer-Pinto M., Bugnot A.B., Johnston E.L. & Dafforn K.A. (2019) Eco-engineering increases habitat availability and utilisation of seawalls by fish. *Ecological Engineering*, 138, 403–411.
(7) Bradford T.E., Astudillo J.C., Lau E.T.C., Perkins M.J., Lo C.C., Li T.C.H., Lam C.S., Ng T.P.T., Strain E.M.A., Steinberg P.D. & Leung K.M.Y. (2020) Provision of refugia and seeding with native bivalves can enhance biodiversity on vertical seawalls. *Marine Pollution Bulletin*, 160, 111578.
(8) Heery E.C., Lian K.Y., Loke L.H.L., Tan H.T.W. & Todd P.A. (2020) Evaluating seaweed farming as an eco-engineering strategy for 'blue' shoreline infrastructure. *Ecological Engineering*, 152, 105857.
(9) Strain E.M.A., Cumbo V.R., Morris R.L., Steinberg P.D. & Bishop M.J. (2020) Interacting effects of habitat structure and seeding with oysters on the intertidal biodiversity of seawalls. *PLoS ONE*, 15, e0230807.
(10) Vozzo M.L., Mayer-Pinto M., Bishop M.J., Cumbo V.R., Bugnot A.B., Dafforn K.A., Johnston E.L., Steinberg P.D. & Strain E.M.A. (2021) Making seawalls multifunctional: the positive effects of seeded bivalves and habitat structure on species diversity and filtration rates. *Marine Environmental Research*, 165, 105243.

2.20. Control or remove non-native or nuisance species on intertidal artificial structures

- We found no studies that evaluated the effects of controlling or removing non-native or nuisance species on intertidal artificial structures on the biodiversity of those structures.

This means we did not find any studies that directly evaluated this intervention during our literature searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Background

Marine artificial structures often support non-native and nuisance species (Airoldi *et al.* 2015; Dafforn 2017), especially those built in urban areas with high vessel movement, disturbance from human activity and poor water quality (Airoldi & Bulleri 2011; Mineur *et al.* 2012). This can have negative effects on native marine biodiversity on and around structures, as well as on humans.

It may be possible to control or remove non-native or nuisance species on intertidal artificial structures to enhance their biodiversity, using mechanical, chemical or biological methods. However, careful consideration must be given to what constitutes a non-native or nuisance species in any given location and scenario, to warrant its control or removal. In this synopsis, species are considered non-native when considered-so in the original study. However, 'nuisance' species that are not also non-native only includes those that have a negative effect on native biodiversity (e.g. by dominating space or smothering) – *not* those that are only a nuisance to society (e.g. by creating slippery surfaces, overgrowing aquaculture species, or fouling infrastructure). Care must also be taken to avoid causing unintended harm to non-target organisms (Locke *et al.* 2009).

Studies investigating control/removal actions that are indiscriminate and simultaneously remove all biodiversity from structure surfaces (e.g. Novak *et al.* 2017) or aim to prevent or reduce colonization in the first place for the benefit of humans (i.e. biofouling reduction; Scardino & de Nys 2011) are not included in this synopsis, which focusses on actions to enhance the biodiversity of artificial structures. Studies that only report the effects of actions on the controlled/removed species itself and not on the wider native biodiversity of structures are not included. Studies that report the effects of patch-scale control/removal, where continued presence on surrounding surfaces would be expected to influence conservation outcomes in practice, are not included but are informative (e.g. Dumont *et al.* 2011). Studies that investigate the effects of actions associated with maintenance or harvesting activities that reduce the likelihood of non-native species occupying bare space made available on structure surfaces following these activities are not considered here, but are included under "*Cease or alter maintenance activities on intertidal artificial structures*" and "*Manage or restrict harvesting of species on intertidal artificial structures*".

Definition: 'Controlling or removing non-native or nuisance species' includes actions taken to reduce the abundance of non-native or nuisance organisms on structures with the aim of enhancing their biodiversity.

See also: *Cease or alter maintenance activities on intertidal artificial structures*; *Manage or restrict harvesting of species on intertidal artificial structures*.

Airoldi L. & Bulleri F. (2011) Anthropogenic disturbance can determine the magnitude of opportunistic species responses on marine urban infrastructures. *PLoS ONE*, 6, e22985.

Airoldi L., Turon X., Perkol-Finkel S. & Rius M. (2015) Corridors for aliens but not for natives: effects of marine urban sprawl at a regional scale. *Diversity and Distributions*, 21, 755–768.

Dafforn K.A. (2017) Eco-engineering and management strategies for marine infrastructures to reduce establishment and dispersal of non-indigenous species. *Management of Biological Invasions*, 8, 153–161.

Dumont C.P., Harris L.G. & Gaymer C.F. (2011) Anthropogenic structures as a spatial refuge from predation for the invasive bryozoan *Bugula neritina*. *Marine Ecology Progress Series*, 427, 95–103.

Locke A., Doe K.G., Fairchild W.L., Jackman P.M. & Reese E.J. (2009) Preliminary evaluation of effects of invasive tunicate management with acetic acid and calcium hydroxide on non-target marine organisms in Prince Edward Island, Canada. *Aquatic Invasions*, 4, 221–236.

Mineur F., Cook E.J., Minchin D., Bohn K., Macleod A. & Maggs C.A. (2012) Changing coasts: marine aliens and artificial structures. *Oceanography and Marine Biology: An Annual Review*, 50, 189–234.

Novak L., López-Legentil S., Sieradzki E. & Shenkar N. (2017) Rapid establishment of the non-indigenous ascidian *Styela plicata* and its associated bacteria in marinas and fishing harbors along the Mediterranean coast of Israel. *Mediterranean Marine Science*, 18, 324–331.

Scardino A.J. & de Nys R. (2011) Mini review: biomimetic models and bioinspired surfaces for fouling control. *Biofouling: The Journal of Bioadhesion and Biofilm Research*, 27, 73–86.

2.21. Cease or alter maintenance activities on intertidal artificial structures

- We found no studies that evaluated the effects of ceasing or altering maintenance activities on intertidal artificial structures on the biodiversity of those structures.

This means we did not find any studies that directly evaluated this intervention during our literature searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Background

Intertidal rocky habitats experience intermittent disturbance from storms, sedimentation, pollution and other human activities, which lead to fluctuations in biodiversity (e.g. Vaselli *et al.* 2008). These pressures are often more pronounced and frequent on artificial structures, especially those built in urban areas with high human activity and poor water quality, and/or in areas of high wave energy (Airoldi & Bulleri 2011; Moschella *et al.* 2005). Artificial structures are also often subject to disturbance from maintenance activities carried out to ensure they remain fit-for-purpose, safe, and aesthetically acceptable. Maintenance can include repairing or reinforcing points of weakness such as eroded cracks or holes, moving or replacing dislodged components, or cleaning regimes using physical or chemical methods. Such activities can further disturb, damage or remove biodiversity from structure surfaces (Airoldi & Bulleri 2011; Sherrard *et al.* 2016), reduce the availability of microhabitats for organisms to shelter in (Moreira *et al.* 2007), and leave bare space available to opportunistic non-native or other nuisance species (Bulleri & Airoldi 2005).

Although some maintenance is likely to be essential, there may be opportunities to cease or alter activities that disturb, damage or remove native organisms from intertidal artificial structures, to maintain or enhance their biodiversity. Altering activities could include using lower-impact methods, reducing the frequency or adjusting the timing of maintenance to avoid disturbance, damage, removal or the creation of bare space on surfaces when non-native or problematic species are more likely to occupy it (Airoldi & Bulleri 2011; Bulleri & Airoldi 2005; Gallagher *et al.* 2016). It could also include allowing natural weathering of structure surfaces to occur, creating rough texture and microhabitat spaces (Moreira *et al.* 2007).

Studies of the effects of real or simulated maintenance activities to illustrate their impact compared with no or altered maintenance, where it is not clear that ceasing/altering maintenance would be a feasible conservation action, are not included but are informative (e.g. Airoldi & Bulleri 2011; Bulleri & Airoldi 2005; Moreira *et al.* 2007). Studies that investigate cleaning activities to control or remove non-native or nuisance species are similarly not included where these actions are indiscriminate, simultaneously removing all biodiversity (e.g. Novak *et al.* 2017).

Definition: 'Cease or alter maintenance activities' includes actions taken to avoid or reduce the disturbance, damage or removal of native organisms from structures, with the aim of enhancing their biodiversity.

See also: *Control or remove non-native or nuisance species on intertidal artificial structures*; *Manage or restrict harvesting of species on intertidal artificial structures*.

Airoldi L. & Bulleri F. (2011) Anthropogenic disturbance can determine the magnitude of opportunistic species responses on marine urban infrastructures. *PLoS ONE*, 6, e22985.

Bulleri F. & Airoldi L. (2005) Artificial marine structures facilitate the spread of a non-indigenous green alga, *Codium fragile* ssp. *tomentosoides*, in the north Adriatic Sea. *Journal of Applied Ecology*, 42, 1063–1072.

Gallagher M.C., Culloty S., McAllen R. & O'Riordan R. (2016) Room for one more? Coexistence of native and non-indigenous barnacle species. *Biological Invasions*, 18, 3033–3046.

Moreira J., Chapman M.G. & Underwood A.J. (2007) Maintenance of chitons on seawalls using crevices on sandstone blocks as habitat in Sydney Harbour, Australia. *Journal of Experimental Marine Biology and Ecology*, 347, 134–143.

Moschella P.S., Abbiati M., Åberg P., Airoldi L., Anderson J.M., Bacchiocchi F., Bulleri F., Dinesen G.E., Frost M., Gacia E., Granhag L., Jonsson P.R., Satta M.P., Sundelöf A., Thompson R.C. & Hawkins S.J. (2005) Low-crested coastal defence structures as artificial habitats for marine life: using ecological criteria in design. *Coastal Engineering*, 52, 1053–1071.

Novak L., López-Legentil S., Sieradzki E. & Shenkar N. (2017) Rapid establishment of the non-indigenous ascidian *Styela plicata* and its associated bacteria in marinas and fishing harbors along the Mediterranean coast of Israel. *Mediterranean Marine Science*, 18, 324–331.

Sherrard T.R.W., Hawkins S.J., Barfield P., Kitou M., Bray S. & Osborne P.E. (2016) Hidden biodiversity in cryptic habitats provided by porous coastal defence structures. *Coastal Engineering*, 118, 12–20.

Vaselli S., Bertocci I., Maggi E. & Benedetti-Cecchi L. (2008) Effects of mean intensity and temporal variance of sediment scouring events on assemblages of rocky shores. *Marine Ecology Progress Series*, 364, 57–66.

2.22. Manage or restrict harvesting of species on intertidal artificial structures

- **Two studies** examined the effects of managing or restricting harvesting of species on intertidal artificial structures on the biodiversity of those structures or on human behaviour likely to influence the biodiversity of those structures. One study was on open coastlines in Italy[1], and one was in ports and on open coastlines in Gibraltar and southeast Spain[2].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (1 STUDY)

- **Invertebrate abundance (1 study):** One replicated, site comparison study in Gibraltar and Spain[2] reported that restricting human access on intertidal artificial structures did not increase the limpet abundance on structure surfaces.
- **Invertebrate condition (1 study):** One replicated, site comparison study in Gibraltar and Spain[2] found that restricting human access on intertidal artificial structures resulted in larger limpets with more balanced sex ratios than unrestricted structures.

BEHAVIOUR (1 STUDY)

- **Human behaviour change (1 study):** One replicated, randomized study in Italy[1] reported that legally restricting human access on intertidal artificial structures did not prevent people from harvesting invertebrates and fishes on and around structures.

Background

Intertidal rocky habitats experience intermittent disturbance from storms, sedimentation, pollution and other human activities, which lead to fluctuations in biodiversity (e.g. Vaselli *et al.* 2008). These pressures are often more pronounced and frequent on artificial structures, especially those built in urban areas with high human activity and poor water quality, and/or in areas of high wave energy (Airoldi & Bulleri 2011; Moschella *et al.* 2005). Artificial structures can also be subject to disturbance from recreational or commercial harvesting of species for food, bait, recreation or souvenirs. Such activities can further disturb, damage or remove biodiversity from structure surfaces (Bulleri & Airoldi 2005; Airoldi *et al.* 2005) and can leave bare space available to opportunistic non-native or other nuisance species (Bulleri & Airoldi 2005).

In some circumstances, it may be desirable for structures to support multifunctional recreational or commercial activities, potentially diverting pressure away from natural habitats (Evans *et al.* 2017). If this is not the case, there may be opportunities to manage or restrict harvesting activities that disturb, damage or remove native organisms from intertidal artificial structures, to maintain or enhance their biodiversity. This could include introducing voluntary or enforced spatial or temporal restrictions, promoting sustainable alternatives, educating harvesters on the impacts of their activities, or a variety of other actions that may alter harvesting behaviour with the aim of enhancing the biodiversity of structures. Some artificial structures already have restricted access with existing surveillance. These may offer a means of creating cost-effective "artificial micro-reserves" where historical cultural rights preclude restricting activities in natural habitats (García-Gómez *et al.* 2010).

Studies of the effects of real or simulated harvesting to illustrate its impact compared with no or managed harvesting, where it is not clear that restricting/managing harvesting would be a feasible conservation action, are not included but are informative (e.g. Airoldi *et al.* 2005; Bulleri & Airoldi 2005).

Definition: 'Managing or restricting harvesting of species' includes actions taken to avoid or reduce the disturbance, damage or removal of native organisms from structures, with the aim of enhancing their biodiversity.

See also: *Control or remove non-native or nuisance species on intertidal artificial structures*; *Cease or alter maintenance activities on intertidal artificial structures*.

Airoldi L., Bacchiocchi F., Cagliola C., Bulleri F. & Abbiati M. (2005) Impact of recreational harvesting on assemblages in artificial rocky habitats. *Marine Ecology Progress Series*, 299, 55–66.

Airoldi L. & Bulleri F. (2011) Anthropogenic disturbance can determine the magnitude of opportunistic species responses on marine urban infrastructures. *PLoS ONE*, 6, e22985.
Bulleri F. & Airoldi L. (2005) Artificial marine structures facilitate the spread of a non-indigenous green alga, *Codium fragile* ssp. *tomentosoides*, in the north Adriatic Sea. *Journal of Applied Ecology*, 42, 1063–1072.
Evans A.J., Garrod B., Firth L.B., Hawkins S.J., Morris-Webb E.S., Goudge H. & Moore P.J. (2017) Stakeholder priorities for multi-functional coastal defence developments and steps to effective implementation. *Marine Policy*, 75, 143–155.
García-Gómez J.C., López-Fé C.M., Espinosa F., Guerra-García J.M. & Rivera-Ingraham G.A. (2010) Marine artificial micro-reserves: a possibility for the conservation of endangered species living on artificial substrata. *Marine Ecology*, 32, 6–14.
Moschella P.S., Abbiati M., Åberg P., Airoldi L., Anderson J.M., Bacchiocchi F., Bulleri F., Dinesen G.E., Frost M., Gacia E., Granhag L., Jonsson P.R., Satta M.P., Sundelöf A., Thompson R.C. & Hawkins S.J. (2005) Low-crested coastal defence structures as artificial habitats for marine life: using ecological criteria in design. *Coastal Engineering*, 52, 1053–1071.
Vaselli S., Bertocci I., Maggi E. & Benedetti-Cecchi L. (2008) Effects of mean intensity and temporal variance of sediment scouring events on assemblages of rocky shores. *Marine Ecology Progress Series*, 364, 57–66.

A replicated, randomized study in 2001–2002 on intertidal breakwaters and groynes in five sites on open coastline in the Adriatic Sea, Italy (1) reported that making access to the breakwaters illegal did not prevent people from harvesting invertebrates and fishes on and around them. At four sites, an average of 0–2 harvesters/2-hour survey were recorded on breakwaters, despite access being illegal. At one site where breakwaters (access illegal) and groynes (access legal) were studied simultaneously, an average of 0–5 harvesters/2-hour survey were recorded. At this site >70% of observations were on groynes, but harvesting also occurred on breakwaters (details not reported). Harvesting species on breakwaters was restricted by making access illegal, but with no apparent enforcement (timing and other details not reported). The number of people harvesting invertebrates and fishes on breakwaters at each of five sites was counted during 2-hour surveys on 152 randomly-selected days between November 2001 and November 2002. Observations at one of the sites included harvesting on groynes, to which access was legal.

A replicated, site comparison study in 2006 on six intertidal breakwaters in ports and on open coastline in the Gibraltar Strait, Gibraltar and the Alboran Sea, Spain (2) found that breakwaters with restricted human access supported similar densities of ribbed Mediterranean limpets *Patella ferruginea* but with larger average size and more balanced sex ratios, compared with unrestricted breakwaters. Limpet density was similar on breakwaters with restricted access (0–7 limpets/m) and those without (3–7/m) (data not statistically tested). On average, limpets were larger on breakwaters with restricted access (4–7 cm) than without (3–4 cm) in seven of nine comparisons, but were similar in two comparisons (both 4 cm). Limpet sex ratio on restricted breakwaters ranged from 2: 1 (males: females) to 18: 1, while the ratio on unrestricted breakwaters ranged from 41: 1 to 117: 1 (data not statistically tested). Harvesting species on breakwaters was restricted by restricting site access. Three breakwaters were in private or military areas with restricted access and surveillance (timing and other details not reported) while three had no access restrictions. Limpet harvesting was technically forbidden at all sites due to its protected species status. Limpets on breakwaters were counted, measured and sexed during low tide during June–August 2006.

(1) Airoldi L., Bacchiocchi F., Cagliola C., Bulleri F. & Abbiati M. (2005) Impact of recreational harvesting on assemblages in artificial rocky habitats. *Marine Ecology Progress Series*, 299, 55–66.
(2) Espinosa F., Rivera-Ingraham G.A., Fa D. & García-Gómez J.C. (2009) Effect of human pressure on population size structures of the endangered ferruginean limpet: toward future management measures. *Journal of Coastal Research*, 25, 857–863.

3. Enhancing the biodiversity of subtidal artificial structures

Background

Subtidal artificial structures (or portions of structures) are permanently covered by seawater even at the lowest state of tide. They include, but are not limited to, support pilings for wind turbines, jetties and other platforms, scour protection and mattresses around pilings, cables or pipelines, seabed foundations and anchor weights, submerged surfaces of floating structures such as pontoons, and also subtidal portions of coastline structures such as seawalls and breakwaters. The names given to different types of structures vary with geography, so we provide a Glossary of terms in Appendix 1 with alternative names for those used here.

Subtidal artificial structures tend to support lower biodiversity than natural subtidal rocky habitats and are often colonized by weedy or opportunistic species, including non-natives (Glasby *et al.* 2006). There are various reasons for their reduced biodiversity. Structures often have steeper inclinations than natural reefs, with narrower bands of subtidal habitat, meaning that space for organisms is scarce and competitive interactions and other environmental processes differ (Chapman & Underwood 2011). The materials used in construction can be unfavourable for certain species (Glasby *et al.* 2006), while the uniform shapes and flat surfaces of many structures offer low habitat complexity with fewer niche spaces (Wilhelmsson & Malm 2008). Furthermore, structure designs can present novel habitat conditions not otherwise found in nature (e.g. sheltered sides of shore-parallel breakwaters built on exposed coastlines; shaded surfaces on pilings under jetties; downward-facing surfaces at constant depth under floating pontoons).

The eco-engineering approaches described here focus on altering structure designs to increase their habitat complexity and niche availability, with the aim of enhancing their biodiversity. They also explore using alternative construction materials, transplanting or seeding species directly onto structures, controlling or removing non-native or nuisance species, and managing human disturbances from maintenance and harvesting activities. It is important to recognize the importance of the environmental context in which structures are placed in influencing the biodiversity that can colonize and survive on them, and thus the likely effects of interventions in different scenarios. Structures built in urban environments may be subject to high human disturbance (Airoldi & Bulleri 2011) and poor water quality (Perrett *et al.* 2006). Those built on exposed sandy or muddy shorelines (i.e where coastal protection or reinforcement is often required) may be frequently disturbed by wave energy and sediment scouring/burial (Moschella *et al.* 2005), and may be too far from source populations for some species to colonize.

It is crucial, therefore, that decision-makers understand the ecology of the species and communities they wish to target with their actions to enhance biodiversity on subtidal artificial structures, and the environmental context in which their structures are located. We encourage the reader to take particular note of the location and context in which studies summarized here were carried out, along with the spatial scale and timeframe over which effects were monitored.

This chapter describes 21 conservation interventions that could be carried out to enhance the biodiversity of subtidal artificial structures or subtidal portions of artificial structures that also extend into the intertidal. We found evidence for the effects of 15 of these interventions. Definitions are provided in the background sections for each intervention (also see intervention list in Appendix 2). These are particularly important for interventions that involve creating artificial habitats or shelters. There has been little consistency in the literature to date in naming conventions for habitats occurring in nature or created for conservation intervention. One person's 'crevice' may be another's 'groove'. Here, we define habitats according to their size and shape. The "See also" sections at the end of each background signpost the reader to similar or related interventions.

Airoldi L. & Bulleri F. (2011) Anthropogenic disturbance can determine the magnitude of opportunistic species responses on marine urban infrastructures. *PLoS ONE*, 6, e22985.

Chapman M.G. & Underwood A.J. (2011) Evaluation of ecological engineering of "armoured" shorelines to improve their value as habitat. *Journal of Experimental Marine Biology and Ecology*, 400, 302–313.

Glasby T.M., Connell S.D., Holloway M.G. & Hewitt C.L. (2006) Nonindigenous biota on artificial structures: could habitat creation facilitate biological invasions? *Marine Biology*, 151, 887–895.

Moschella P.S., Abbiati M., Åberg P., Airoldi L., Anderson J.M., Bacchiocchi F., Bulleri F., Dinesen G.E., Frost M., Gacia E., Granhag L., Jonsson P.R., Satta M.P., Sundelöf A., Thompson R.C. & Hawkins S.J. (2005) Low-crested coastal defence structures as artificial habitats for marine life: using ecological criteria in design. *Coastal Engineering*, 52, 1053–1071.

Perrett L.A., Johnston E.L. & Poore A.G.B. (2006) Impact by association: direct and indirect effects of copper exposure on mobile invertebrate fauna. *Marine Ecology Progress Series*, 326, 195–205.

Wilhelmsson D. & Malm T. (2008) Fouling assemblages on offshore wind power plants and adjacent substrata. *Estuarine, Coastal and Shelf Science*, 79, 459–466.

3.1. Use environmentally-sensitive material on subtidal artificial structures

- **Fourteen studies** examined the effects of using environmentally-sensitive material on subtidal artificial structures on the biodiversity of those structures. Seven studies were on open coastlines in the United Arab Emirates[2], Italy[3,5], Israel[6,7], southeast Spain[12], and in France, the UK, Portugal and Spain[14]. Three were in marinas in northern Israel[9] and the UK[10,11], two were in estuaries in southeast Australia[1] and eastern USA[8], one was in a lagoon in Mayotte[4], and one was in a port in Germany[13].

COMMUNITY RESPONSE (11 STUDIES)

- **Overall community composition (11 studies):** Six of 11 replicated, controlled studies (including eight randomized, three paired sites and one before-and-after study) in Australia[1], the United Arab Emirates[2], Italy[3], Israel[6,7,9], the USA[8], the UK[10,11], Spain[12] and Germany[13] found that using shell-concrete[10] or quarried rock[12] in place of standard-concrete on subtidal artificial structures, or using ECOncrete™[6,7,8,9] in place of standard-concrete[6,7,9] or fibreglass[8], along with creating texture[8], grooves[7,9], holes[7,9], pits[7] and/or small ledges[9], altered the combined macroalgae and invertebrate community composition on structure surfaces. Three studies[1,2,13] found that using quarried rock[1,2] or blast-furnace-cement-concrete[13] in place of standard-concrete did not alter the community composition, while one[3] found mixed effects depending on the type of rock tested and the site. One[11] found that using different cement mixes in concrete (including some with recycled cements) altered the community composition of native species, but not non-natives. Three of the studies[7,8,9] also reported that ECOncrete™ surfaces with added habitats supported mobile invertebrate[7,8], non-mobile invertebrate[7,8,9] and/or fish[7] species that were absent from standard-concrete[7,9] or fibreglass[8] structure surfaces.

- **Overall richness/diversity (7 studies):** Three of seven replicated, controlled studies (including five randomized, two paired sites and one before-and-after study) in Italy[3], Israel[7,9], the USA[8], the UK[10,11] and Spain[12] found that using quarried rock[3], shell-concrete[10] or recycled-cement-concrete[11] in place of standard-concrete on subtidal artificial structures had mixed effects on the combined macroalgae and invertebrate species richness on structure surfaces, depending on the site[3], surface orientation[10] or type of cement tested[11]. One of the studies[10], along with one other[12], found that using shell-concrete[10] or quarried rock[12] did not increase the species diversity[10,12] and/or richness[12], while one[11] found that using recycled cement did not increase the non-native species richness. Three studies[7,8,9] found that using ECOncrete™, along with creating texture[8], grooves[7,9], holes[7,9], pits[7] and/or small ledges[9], did increase the species diversity[7,9] and/or richness[8,9] on and around structures.
- **Algal richness/diversity (1 study):** One replicated, randomized, controlled study in the UK[11] found that using recycled-cement-concrete in place of standard-concrete on subtidal artificial structures did not increase the diatom species richness on structure surfaces.

POPULATION RESPONSE (11 STUDIES)

- **Overall abundance (7 studies):** Three of seven replicated studies (including six controlled, four randomized and one paired sites study) in the United Arab Emirates[2], Italy[3], Israel[6], the USA[8], the UK[10], Spain[12], and in France, the UK, Portugal and Spain[14] found that using quarried rock[2,3] or shell-concrete[10] in place of standard-concrete on subtidal artificial structures did not increase the combined macroalgae and invertebrate abundance on structure surfaces. Two studies[12,14] found mixed effects, depending on the type of quarried rock[12] or concrete[14] tested and/or the location[14]. One[8] found that using ECOncrete™ in place of fibreglass, along with creating textured surfaces, increased the live cover and biomass, while one[6] found that different ECOncrete™ and standard-concrete mixes supported different cover and inorganic biomass but similar organic biomass.
- **Algal abundance (6 studies):** Four of six replicated, controlled studies (including four randomized and one paired sites study) in Australia[1], the United Arab Emirates[2], Italy[3,5], Israel[7] and the UK[11] found that using quarried rock[1,2,5] or recycled-cement-concrete[11] in place of standard-concrete on subtidal artificial structures did not increase the abundance of brown[1], turf[2] or coralline[2] macroalgae, canopy macroalgae recruits[5] or diatoms[11] on structure surfaces. Two studies found that using quarried rock[3] or using ECOncrete™, along with creating grooves, holes and pits[7], had mixed effects on macroalgal abundance, depending on the species group[3,7] and/or site[3]. One of the studies[1] found that using quarried rock increased red and green macroalgal abundance.

- **Invertebrate abundance (6 studies):** Three of six replicated, controlled studies (including four randomized and one paired sites study) in Austalia[1], the United Arab Emirates[2], Italy[3], Israel[6,7] and the UK[10] found that using quarried rock[2,3] in place of concrete on subtidal artificial structures, or using ECOncrete™, along with creating grooves, holes and pits[7], had mixed effects on the abundance of non-mobile invertebrates[3,7], mobile invertebrates[7] or coral recruits[2] on structure surfaces, depending on the type of rock tested[2], the species group[7] and/or the site[2,3]. One of the studies[2], along with one other[1], found that using quarried rock did not increase the abundance of sponges[1,2], bryozoans[1,2], ascidians[1,2], mussels[1], barnacles[1], or Serpulid tubeworms[1], but in one[1] it decreased Spirorbid tubeworm abundance. One study[10] found that using shell-concrete increased bivalve abundance. One[6] found that different ECOncrete™ and standard-concrete mixes supported different coral abundance.
- **Fish abundance (1 study):** One replicated, controlled study in Israel[7] found that using ECOncrete™ in place of standard-concrete on subtidal artificial structures, along with creating grooves, holes and pits, had mixed effects on fish abundances, depending on the species group.

BEHAVIOUR (1 STUDY)

- **Use (1 study):** One study in Mayotte[4] reported that basalt rock surfaces created on a concrete subtidal artificial structure, along with small and large swimthroughs, were used by juvenile spiny lobsters and groupers, sea firs, and adult fishes from five families.

Background

Material type influences the settlement and survival of marine organisms in subtidal rocky habitats. Settlement preferences and competition lead to some species being more abundant than others on certain materials (Andersson *et al.* 2009; Guidetti *et al.* 2004), but patterns vary by environmental conditions. Physical (lithology, hardness, porosity, colour, texture) and chemical (pH, mineralogy, toxicity) properties of rock and manufactured materials can affect how they weather over time and what communities develop on them (Coombes *et al.* 2011).

Marine artificial structures are often made from hard quarried rock, concrete, wood, steel or plastic, according to engineering requirements, cost and/or availability. Synthetic materials can be associated with the presence of non-native species (Dafforn 2017), whereas structures made from natural rock may support more natural rocky reef communities. There may be opportunities to use more environmentally-sensitive materials in structures or in eco-engineering habitat designs added to structures to enhance their biodiversity. Concrete is commonly-used in eco-engineering designs since it is durable and easy to mould into complex shapes. Yet adding manufactured concrete habitats to structures to enhance biodiversity may not deliver a net environmental gain because of the large CO_2 footprint of concrete production (Heery *et al.* 2020). Concrete mixes can be manipulated to alter their physical and chemical properties (McManus *et al.* 2018; Natanzi *et al.* 2021) and environmental footprint (Dennis *et al.* 2018). Lower-footprint materials may be preferable, regardless of their effect on colonizing biodiversity; a neutral/no effect on biodiversity may still offer a higher net environmental gain (or lower net loss).

It is often not possible to separate the effects of the various physical and chemical properties of materials on biodiversity. Studies that directly examine the effects of creating different surface textures are included under the action "*Create textured surfaces (≤1 mm) on subtidal artificial structures*"; any other material comparisons are considered here. There are bodies of literature investigating the effects of material on settlement behaviour and ecological interactions in the laboratory and field (e.g. Davis *et al.* 2017; Glasby 2000), for artificial reefs and coral gardening (reviewed by Baine 2001; Spieler *et al.* 2001), and for anti-fouling applications (e.g. Hanson & Bell 1976). Unless the relevance is explicitly highlighted by the authors, these studies are not included in this synopsis, which focusses on *in situ* conservation actions to promote colonization of biodiversity on marine artificial structures that are engineered to fulfil a primary function other than providing artificial habitats.

Definition: 'Environmentally-sensitive materials' are materials that seek to maximize environmental benefits and/or minimize environmental risks of marine engineering.

See also: *Create textured surfaces (≤1 mm) on subtidal artificial structures*.

Andersson M.H., Berggren M., Wilhelmsson D. & Öhman M.C. (2009) Epibenthic colonization of concrete and steel pilings in a cold-temperate embayment: a field experiment. *Helgoland Marine Research*, 63, 249–260.

Baine M. (2001) Artificial reefs: a review of their design, application, management and performance. *Ocean & Coastal Management*, 44, 241–259.

Coombes M.A., Naylor L.A., Thompson R.C., Roast S.D., Gómez-Pujol L. & Fairhurst R.J. (2011) Colonization and weathering of engineering materials by marine microorganisms: an SEM study. *Earth Surface Processes and Lanforms*, 36, 585–593.

Dafforn K.A. (2017) Eco-engineering and management strategies for marine infrastructures to reduce establishment and dispersal of non-indigenous species. *Management of Biological Invasions*, 8, 153–161.

Davis K.L., Coleman M.A., Connell S.D., Russell B.D., Gillanders B.M. & Kelaher B.P. (2017) Ecological performance of construction materials subject to ocean climate change. *Marine Environmental Research*, 131, 177–182.

Dennis H.D., Evans A.J., Banner A.J. & Moore P.J. (2018) Reefcrete: reducing the environmental footprint of concretes for eco-engineering marine structures. *Ecological Engineering*, 120, 668–678.

Glasby T.M. (2000) Surface composition and orientation interact to affect subtidal epibiota. *Journal of Experimental Marine Biology and Ecology*, 248, 177–190.

Guidetti P., Bianchi C.N., Chiantore M., Schiaparelli S., Morri C. & Cattaneo-Vietti R. (2004) Living on the rocks: substrate mineralogy and the structure of subtidal rocky substrate communities in the Mediterranean Sea. *Marine Ecology Progress Series*, 274, 57–68.

Hanson C.H. & Bell J. (1976) Subtidal and intertidal marine fouling on artificial substrata in northern Puget Sound, Washington. *Fishery Bulletin* (United States), 74, 2.

Heery E.C., Lian K.Y., Loke L.H.L., Tan H.T.W. & Todd P.A. (2020) Evaluating seaweed farming as an eco-engineering strategy for 'blue' shoreline infrastructure. *Ecological Engineering*, 152, 105857.

McManus R.S., Archibald N., Comber S., Knights A.M., Thompson R.C. & Firth L.B. (2018) Partial replacement of cement for waste aggregates in concrete coastal and marine infrastructure: a foundation for ecological enhancement? *Ecological Engineering*, 120, 655–667.

Natanzi A.S., Thompson B.J., Brooks P.R., Crowe T.P. & McNally C. (2021) Influence of concrete properties on the initial biological colonisation of marine artificial structures. *Ecological Engineering*, 159, 106104.

Spieler R.E., Gilliam D.S. & Sherman R.L. (2001) Artificial substrate and coral reef restoration: what do we need to know to know what we need. *Bulletin of Marine Science*, 69, 1013–1030.

A replicated, randomized, paired sites, controlled study in 1998–1999 on four subtidal pontoons and four rocky reefs in Sydney Harbour estuary, Australia (1) found that sandstone and concrete settlement plates supported similar macroalgae and invertebrate community composition, while abundances varied depending on the species group. After seven months, macroalgae and non-mobile invertebrate community composition was similar on sandstone and concrete settlement plates (data reported as statistical model results). Sandstone plates supported higher abundance of red macroalgae (2–33% cover) and green macroalgae (2–13%) than concrete plates (red: 1–15%; green: 1–7%), but fewer Spirorbid tubeworms (0–37 vs 0–48%). Abundances were similar on sandstone and concrete plates for brown macroalgae (1–10 vs 1–17%), mussels (*Mytilus edulis*: 1–34 vs 3–44%), barnacles (Cirripedia: 1–37 vs 1–41%), sponges (Porifera: both 1–9%), bryozoans (Bryozoa: 0–38 vs 0–49%), ascidians (Ascidiacea: 0–25 vs 0–16%) and Serpulid tubeworms (3–17 vs 3–16%). Settlement plates (150 × 150 mm) were made from sandstone and concrete. Five of each were randomly arranged vertically at 0.3 m depth on each of four concrete pontoons and at 1.5 m depth on each of four adjacent sandstone reefs in June 1998. Macroalgae and non-mobile invertebrates on plates were counted in the laboratory after seven months.

A replicated, randomized, controlled study in 2007–2008 on two subtidal breakwaters and two rocky reefs on open coastline in the Persian Gulf, United Arab Emirates (2) found that sandstone, terracotta, granite, gabbro and concrete settlement plates supported similar macroalgae and invertebrate community composition and abundances, while juvenile coral (Scleractinia, Alcyonacea) abundances varied depending on the site. After 12 months, macroalgae and non-mobile invertebrate cover was similar on sandstone (65%), terracotta (75%), granite (85%), gabbro (80%) and concrete (79%) settlement plates. The same was true for abundances of macroalgal turf, coralline algae (Corallinales), sponges (Porifera), bryozoans (Bryozoa) and ascidians (Ascidiacea) (data not reported) and the community composition (data reported as statistical model results). At one of four sites, juvenile corals were more abundant on gabbro (7 colonies/plate) than sandstone (3/plate) and concrete (3/plate) plates, while no other significant differences were found (terracotta: 7/plate; granite: 5/plate). At the other sites, few corals were recorded with no significant differences between materials (all 0/plate). Settlement plates (100 × 100 mm) were made from sandstone, terracotta, granite, gabbro and concrete. Twenty-five of each were randomly arranged horizontally 10–15 mm above the substrate at 4 m depth on each of two breakwaters and two rocky reefs in April 2007. Macroalgae and non-mobile invertebrates on the undersides of plates were counted from photographs and juvenile corals in the laboratory after 12 months. Twenty-five plates were missing and no longer provided habitat.

A replicated, randomized, controlled study in 2005 on three subtidal rocky reefs on open coastlines in the Adriatic Sea and the Ionian Sea, Italy (3) found that limestone, sandstone, granite and concrete settlement plates supported similar macroalgae and non-mobile invertebrate live cover, while community composition, species richness and abundances varied depending on the site and species group. After nine months, macroalgae and non-mobile invertebrate community composition differed between limestone and concrete and between sandstone and concrete plates in four of six sites each, and between sandstone and concrete plates in two of six sites, but did not differ in the other sites (data reported as statistical model results). Live cover was similar on all materials, while species richness comparisons varied by site (data not reported). Abundance comparisons varied by species group and site (see paper for results). Limestone, sandstone, granite and concrete settlement plates (150 × 100 mm) were made with and without textured surfaces. Five of each material-texture combination were randomly arranged horizontally at 5 m depth in each of two sites on each of three limestone reefs in February 2005. Macroalgae and non-mobile invertebrates on plates were counted in the laboratory after nine months.

A study in 2009–2010 on a subtidal pipeline in a lagoon in the Mozambique Channel, Mayotte (4) reported that pipeline anchor-weights with basalt rock surfaces created on them, along with small and large swimthroughs, were used by juvenile spiny lobsters *Panulirus versicolor*, juvenile blue-and-yellow groupers *Epinephelus flavocaeruleus*, sea firs (Hydrozoa), and adult fishes from five families. After one month, juvenile spiny lobsters and blue-and-yellow groupers, sea firs, and adult damselfish/clownfish (Pomacentridae), wrasse (Labridae), butterflyfish (Chaetodontidae), squirrelfish/soldierfish (Holocentridae) and surgeonfish (Acanthuridae) were recorded on and around anchor-weights with basalt rocks and swimthrough habitats. Basalt rocks (dimensions/numbers not reported) were attached over horizontal surfaces of concrete anchor-weights placed over a seabed pipeline (400 mm diameter). Small and large swimthrough habitats were also created on the anchor-weights. A total of 260 anchor-weights were placed with one every 10 m along the pipeline at 0–26 m depth during December 2009–March 2010. Fishes were counted on and around the pipeline from videos after 1 month.

A replicated, randomized, controlled study in 2009 on a subtidal rocky reef on open coastline in the Adriatic Sea, Italy (5) found that limestone, clay and concrete settlement plates supported similar numbers of juvenile canopy algae *Cystoseira barbata*. After three months, there was no significant difference in the average number of canopy algae recruits on limestone (25/plate), clay (29/plate) and concrete (12/plate) settlement plates. Six settlement plates (100 × 100 mm) of each of three materials (limestone, clay, concrete) were randomly arranged horizontally on a rocky reef at 3 m depth in March 2009. Recruits of juvenile canopy algae settled onto plates were counted after three months.

A replicated, randomized, controlled study (year not reported) on open coastlines in the Mediterranean Sea and the Gulf of Aqaba, Israel (6) found that settlement plates made from different concrete mixes (including five ECOncrete™ and one standard-concrete mix) supported different macroalgae and non-mobile invertebrate community composition, coral (Scleractinia, Alcyonacea) abundance and inorganic biomass, but similar organic biomass. Over 12 months, macroalgae and non-mobile invertebrate community composition differed on different concrete mixes (including five ECOncrete™ and one standard-concrete mix), but it was not clear which materials differed in which locations (data reported as statistical model results). The same was true for their abundance (ECOncrete™: 81–100% cover; standard-concrete: 80–92%), inorganic biomass (ECOncrete™: 153–659 g/m^2; standard-concrete: 168–332 g/m^2) and coral abundance (2–16 vs 3–5 recruits in total). Organic biomass was similar on ECOncrete™ (16–73 g/m^2) and standard-concrete (30–79 g/m^2). Settlement plates (150 × 150 mm) were moulded from six concrete mixes. Five were patented ECOncrete™ mixes with reduced pH (pH 9–11), reduced Portland-cement and alternative cements and additives (details not reported) while one was standard-concrete (pH 13–14, Portland cement). Plates had textured surfaces on one side and were flat on the other. Ten of each material were randomly arranged horizontally with textured surfaces facing upwards on frames at 6 m depth in the Mediterranean Sea and at 10 m in the Gulf of Aqaba (month/year not reported). Macroalgae and invertebrates on plates were counted and biomass (dry weight) was recorded in the laboratory over 12 months.

A replicated, controlled study in 2012–2014 on two subtidal breakwaters on open coastline in the Mediterranean Sea, Israel (7) found that breakwater blocks made from ECOncrete™, along with pits, grooves and holes created on them, supported different macroalgae and invertebrate community composition with higher species diversity than standard-concrete blocks without added habitats, while macroalgae, invertebrate and fish abundances varied depending on the species group. After 24 months, the macroalgae and invertebrate species diversity was higher on ECOncrete™ blocks with added habitats than standard-concrete blocks without (data reported as Shannon index) and the community composition differed (data reported as statistical model results). Thirty species (7 mobile invertebrates, 14 non-mobile invertebrates, 9 fishes) recorded on and around ECOncrete™ blocks were absent from standard blocks. Species abundances varied on materials depending on the species group (see paper for results). It is not clear whether these effects were the direct result of using environmentally-sensitive material or creating grooves, pits and/or holes. Breakwater blocks (1 × 1 × 1 m) were made from three patented ECOncrete™ materials (lower pH and different cement/additives to standard concrete) using a formliner. Five of each were placed at 5–7 m depth on a concrete-block breakwater during construction in July 2012. Blocks had multiple grooves, pits and holes. Five standard-concrete blocks (1.7 × 1.7 × 1.7 m) without added habitats were placed on a similar breakwater 80 m away. Macroalgae and invertebrates on blocks, and fishes on and around blocks, were counted over 24 months.

A replicated, controlled study in 2013–2014 on 24 jetty pilings in the Hudson River estuary, USA (8) found that using ECOncrete™ on pilings, along with creating textured surfaces, increased the macroalgae and invertebrate species richness, cover and biomass and altered the community composition on piling surfaces. After 14 months, ECOncrete™ pilings with textured surfaces supported 18 macroalgae and invertebrate species with 90–100% cover, while fibreglass pilings without texture supported nine species with 40–85% cover (data not statistically tested). Biomass was higher on ECOncrete™ pilings (0.07 g/cm^2) than fibreglass pilings (0.02 g/cm^2) and the community composition differed (data reported as statistical model results). Over 14 months, six species (4 non-mobile invertebrates, 2 mobile invertebrates) recorded on ECOncrete™ pilings were absent from fibreglass ones. It is not clear whether these effects were the direct result of using environmentally-sensitive material or creating textured surfaces. Jetty piling encasements were made from patented ECOncrete™ material using a formliner during maintenance works. Nine ECOncrete™ encasements with textured surfaces and three untextured fibreglass encasements were attached around pilings in each of two sites on a jetty in June 2013 (depth not reported). Macroalgae and invertebrates were counted on and around pilings and biomass was measured (dry weight) in the laboratory over 14 months.

A replicated, randomized, paired sites, controlled, before-and-after study in 2014–2016 on a subtidal seawall in a marina in the Mediterranean Sea, Israel (9) found that seawall panels made from ECOncrete™, along with grooves, small ledges and holes created on them, supported higher macroalgae and invertebrate species diversity and richness and different community composition compared with standard-concrete seawall surfaces without added habitats. After 22 months, macroalgae and invertebrate species diversity (data reported as Shannon index) and richness was higher on ECOncrete™ panels with added habitats (9 species/quadrat) than on standard-concrete seawall surfaces without (5/quadrat), and compared with seawall surfaces before panels were attached (1/quadrat). Community composition differed between ECOncrete™ panels and standard-concrete surfaces (data reported as statistical model results). Two non-mobile invertebrate species groups recorded on panels were absent from standard-concrete surfaces. It is not clear whether these effects were the direct result of using environmentally-sensitive material or creating grooves, ledges and/or holes. Seawall panels (height: 1.5 m; width: 0.9 m; thickness: 130 mm) were made from patented ECOncrete™ material using a formliner. Panels had multiple grooves, small ledges and holes. Four panels were attached to a vertical concrete seawall in November 2014. The bottom 1.2 m were subtidal. Panels were compared with standard-concrete seawall surfaces cleared of organisms (height: 1.2 m; width: 0.9 m) adjacent to each panel. Macroalgae and invertebrates were counted in one 300 × 300 mm randomly-placed quadrat on each panel and seawall surface during high tide over 22 months.

A replicated, randomized, controlled study in 2016 in a marina in the Fal estuary, UK (10) found that shell-concrete settlement plates supported different macroalgae and invertebrate community composition, with higher bivalve abundance but similar species diversity and live cover to standard-concrete plates, while species richness varied depending on the surface orientation. After six months, shell-concrete settlement plates supported different macroalgae and invertebrate community composition (data reported as statistical model results) with similar species diversity and live cover (data not reported) to standard-concrete plates. Species richness comparisons varied depending on the surface orientation (data not reported). Bivalve abundance (*Anomia ephippium*, *Hiatella arctica*, *Musculus costulatas*) was 38% higher on shell-concrete than standard-concrete plates. Settlement plates (150 × 150 mm) were moulded from oyster-shell-concrete and standard-concrete. Plates had grooves and small protrusions on one surface, but were flat on the other. Forty plates were suspended horizontally, randomly arranged, beneath floating pontoons at 2–3 m depth in April 2016. Ten of each material had grooves/protrusions facing up, while 10 of each faced down. Macroalgae and invertebrates on upward- and downward-facing surfaces were counted in the laboratory over six months.

A replicated, randomized, controlled study in 2016 in a marina in the Plym estuary, UK (11) found that replacing standard Portland-cement with Ground Granulated Blast-Furnace Slag (GGBS), Pulverized Fly Ash (PFA), or a mix of both, in concrete settlement plates did not affect the diatom species richness or abundance on plates, or the non-native macroalgae and non-mobile invertebrate species richness or community composition, but had mixed effects on the native species richness and community composition, depending on the cement used. Over four weeks, diatom species richness and live cover was similar on GGBS-concrete (2 species/plate; 19% cover), PFA-concrete (2/plate; 12%), mixed-concrete (2/plate; 20%) and standard-concrete (2/plate; 12%) settlement plates. After seven weeks, native macroalgae and non-mobile invertebrate community composition differed on different materials (data reported as statistical model results), but it was not clear which materials differed. Native species richness was similar on PFA-concrete (8 species/plate) and standard-concrete (9/plate), but lower on GGBS-concrete (8/plate) and mixed-concrete (7/plate) than standard-concrete. Non-native community composition and species richness was similar on all materials (1–2 species/plate). Settlement plates (20 × 20 mm) were moulded with recycled cement (GGBS, PFA, or a mix of both) or standard Portland-cement. Eighty of each were suspended vertically, randomly arranged, beneath floating pontoons at 0.5 m depth in June 2016. Diatoms on plates were counted using a scanning electron microscope over four weeks, and macroalgae and invertebrates in the laboratory after seven weeks.

A replicated, randomized, paired sites, controlled study in 2014–2015 on a subtidal rocky reef on open coastline in the Alboran Sea, Spain (12) found that sandstone, limestone, slate and gabbro settlement plates supported similar macroalgae and non-mobile invertebrate species diversity and richness but different community composition to concrete plates, and that live cover was higher on sandstone than concrete plates. Over 11 months, macroalgae and non-mobile invertebrate species diversity was similar on sandstone, limestone, slate, gabbro and concrete settlement plates (data reported as Shannon index). Community composition differed on all materials, apart from sandstone vs gabbro and slate vs gabbro (data reported as statistical model results), and sandstone plates were more similar to natural rock surfaces than the other materials were (data not statistically tested). Total live cover was higher on sandstone than concrete and gabbro plates, while species richness was higher on sandstone than limestone (data not reported), but no other significant differences were found. Settlement plates (170 × 170 mm) were made from sandstone, limestone, slate, gabbro or concrete. Two of each material were randomly arranged horizontally at 15 m depth in each of three sites on a gneiss reef in June 2014. Plate surfaces had grooves and small protrusions. Macroalgae and non-mobile invertebrates on each pair of plates and on adjacent natural rock surfaces (170 × 170 mm) were counted from photographs over 11 months.

A replicated, randomized, controlled study in 2017–2018 in Jade Weser Port in the North Sea, Germany (13) found that using blast-furnace-cement in place of standard Portland-cement and varying the aggregates in concrete settlement blocks did not alter the community composition of macroalgae, microalgae and non-mobile invertebrates on blocks. After 12 months, the community composition of macroalgae, microalgae and non-mobile invertebrates was similar on blast-furnace-cement concrete and standard-concrete blocks regardless of their aggregate composition (data reported as statistical model results). Concrete settlement blocks (150 × 150 × 150 mm) were moulded with blast-furnace cement or standard Portland-cement. There were four blast-furnace-cement concretes with varying aggregate mixes (sand, gravel, metallic slags; see paper for details) and one standard-concrete mix with sand and gravel aggregate. Three blocks of each blast-furnace-cement mix and three standard-concrete blocks were randomly arranged on frames suspended beneath floating pontoons at 1.5 m depth in April 2017. Macroalgae, microalgae and non-mobile invertebrates on top horizontal and both vertical block surfaces were counted in the laboratory after 12 months.

A replicated study (year not reported) on open coastlines in the English Channel, France and the UK, Matosinhos Bay, Portugal, and Santander Bay, Spain (14) reported that concrete mixes with different mortars and recycled aggregates supported different microalgal, macroalgal and invertebrate biomass, depending on the location, but results were not statistically tested. After six months, on average, settlement blocks with geopolymer mortar supported 6 g of algal and invertebrate biomass/block, while blocks with cement mortar supported 7–9 g/block. Biomass was 6–9 g/block with shell-sand aggregate, 6–8 g/block with limestone-sand, and 6–7 g/block with glass-sand. Results varied depending on the location (see paper for location-specific results). Concrete settlement blocks (160 × 40 × 40 mm) were 3D-printed with different mortar (geopolymer, cement) and recycled aggregates (limestone-sand, glass-sand, shell-sand). Nine blocks of each mortar-aggregate combination were attached horizontally to platforms at 1 m depth in each of France, the UK, Portugal and Spain (month/year not reported). Microalgal, macroalgal and invertebrate biomass (dry weight) on blocks was measured in the laboratory over six months.

(1) Connell S.D. (2000) Floating pontoons create novel habitats for subtidal epibiota. *Journal of Experimental Marine Biology and Ecology*, 247, 183–194.

(2) Burt J., Bartholomew A., Bauman A., Saif A. & Sale P.F. (2009) Coral recruitment and early benthic community development on several materials used in the construction of artificial reefs and breakwaters. *Journal of Experimental Marine Biology and Ecology*, 373, 72–78.

(3) Guarnieri G., Terlizzi A., Bevilacqua S. & Fraschetti S. (2009) Local vs regional effects of substratum on early colonization stages of sessile assemblages. *Biofouling: The Journal of Bioadhesion and Biofilm Research*, 25, 593–604.

(4) Pioch S., Saussola P., Kilfoyle K. & Spieler R. (2011) Ecological design of marine construction for socio-economic benefits: ecosystem integration of a pipeline in coral reef area. *Procedia Environmental Sciences*, 9, 148–152.

(5) Perkol-Finkel S., Ferrario F., Nicotera V. & Airoldi, L. (2012) Conservation challenges in urban seascapes: promoting the growth of threatened species on coastal infrastructures. *Journal of Applied Ecology*, 49, 1457–1466.

(6) Perkol-Finkel S. & Sella I. (2014) Ecologically-active concrete for coastal and marine infrastructure: innovative matrices and designs. Pages 1139–1149 in: W. Allsop & K. Burgess (eds.) *From Sea to Shore – Meeting the Challenges of the Sea (Coasts, Marine Structures and Breakwaters 2013)*. ICE publishing.
(7) Sella I. & Perkol-Finkel S. (2015) Blue is the new green – ecological enhancement of concrete based coastal and marine infrastructure. *Ecological Engineering*, 84, 260–272.
(8) Perkol-Finkel S. & Sella I. (2016) Blue is the new green – harnessing urban coastal infrastructure for ecological enhancement. Pages 139–149 in: A. Baptiste (ed.) *Coastal Management: Changing Coast, Changing Climate, Changing Minds*. ICE publishing.
(9) Perkol-Finkel S., Hadary T., Rella A., Shirazi R. & Sella I. (2018) Seascape architecture – incorporating ecological considerations in design of coastal and marine infrastructure. *Ecological Engineering*, 120, 645–654.
(10) Hanlon N., Firth L.B. & Knights A.M. (2018) Time-dependent effects of orientation, heterogeneity and composition determines benthic biological community recruitment patterns on subtidal artificial structures. *Ecological Engineering*, 122, 219–228.
(11) McManus R.S., Archibald N., Comber S., Knights A.M., Thompson R.C. & Firth L.B. (2018) Partial replacement of cement for waste aggregates in concrete coastal and marine infrastructure: a foundation for ecological enhancement? *Ecological Engineering*, 120, 655–667.
(12) Sempere-Valverde J., Ostalé-Valriberas E., Farfán G.M. & Espinosa F. (2018) Substratum type affects recruitment and development of marine assemblages over artificial substrata: a case study in the Alboran Sea. *Estuarine, Coastal and Shelf Science*, 204, 56–65.
(13) Becker L.R., Kröncke I., Ehrenberg A., Feldrappe V. & Bischof K. (2021) Benthic community establishment on different concrete mixtures introduced to a German deep-water port. *Helgoland Marine Research*, 75, 1–12.
(14) Ly O., Yoris-Nobile A.I., Sebaibi N., Blanco-Fernandez E., Boutouil M., Castro-Fresno D., Hall A.E., Herbert R.J.H., Deboucha W., Reis B., Franco J.N., Borges M.T., Sousa-Pinto I., van der Linden P. & Stafford R. (2021) Optimisation of 3D printed concrete for artificial reefs: biofouling and mechanical analysis. *Construction and Building Materials*, 272, 121649.

3.2. Create textured surfaces (≤1 mm) on subtidal artificial structures

- **Three studies** examined the effects of creating textured surfaces on subtidal artificial structures on the biodiversity of those structures. Two studies were on open coastlines in Italy[1] and Israel[2], and one was in an estuary in eastern USA[3].

COMMUNITY RESPONSE (3 STUDIES)

- **Overall community composition (3 studies):** Two of three replicated, controlled studies (including two randomized studies) in Italy[1], Israel[2] and the USA[3] found that creating textured surfaces on subtidal artificial structures, along with using environmentally-sensitive material in one[3], altered the combined macroalgae and invertebrate community composition on structure surfaces[2,3], while one found no effect[1]. One of the studies[3] also reported that textured surfaces with environementally-sensitive material supported mobile and non-mobile invertebrate species that were absent from fibreglass surfaces without texture.
- **Overall richness/diversity (2 studies):** One of two replicated, controlled studies (including one randomized study) in Italy[1] and the USA[3] found that creating textured surfaces on subtidal artificial structures did not increase the combined macroalgae and non-mobile invertebrate species richness on structure surfaces[1]. One study[3] found that creating textured surfaces, along with using environmentally-sensitive material, did.

POPULATION RESPONSE (3 STUDIES)

- **Overall abundance (3 studies):** Two of three replicated, controlled studies (including two randomized studies) in Italy[1], Israel[2] and the USA[3] found that creating textured surfaces on subtidal artificial structures did not increase the combined macroalgae and non-mobile invertebrate live cover on structure surfaces. One study[3] found that creating textured surfaces, along with using environmentally-sensitive material, did increase the cover and biomass.
- **Algal abundance (1 study):** One replicated, randomized, controlled study in Italy[1] found that creating textured surfaces on subtidal artificial structures had mixed effects on the macroalgal abundance on structure surfaces, depending on the species group and site.
- **Invertebrate abundance (1 study):** One replicated, randomized, controlled study in Italy[1] found that creating textured surfaces on subtidal artificial structures had mixed effects on the non-mobile invertebrate abundance on structure surfaces, depending on the site.

BEHAVIOUR (0 STUDIES)

Background

Texture influences the settlement and survival of marine organisms in subtidal rocky habitats. It provides secure anchor points for invertebrate larvae and algal germlings, helping them to resist dislodgement and escape predation or grazing (Carl *et al.* 2012). Settlement preferences and competitive interactions lead to some species being more abundant than others on textured surfaces (Bourget *et al.* 1994). These patterns vary by species, environmental conditions and the match or mismatch between the size and shape of the texture and organisms (Wahl & Hoppe 2002).

Most substrates have some form of texture, but marine artificial structures often have smoother surface texture than natural rocky substrates (Sedano *et al.* 2020). Structures with rougher texture tend to be more-readily colonized by invertebrates and algae (Miller & Barimo 2001; Sempere-Valverde *et al.* 2018; but see Bourget *et al.* 1994), promoting community development. Textured surfaces can be created on subtidal artificial structures by moulding or treating surfaces during construction or retrospectively. Texture can also be altered indirectly through material choice. Studies that examine the effects of using alternative materials with incidentally-different textures are not considered here, but are included under the action "*Use environmentally-sensitive material on subtidal artificial structures*".

There are bodies of literature investigating field and laboratory-based settlement behaviour on textured surfaces (e.g. Maldonado & Uriz 1998; Neo *et al.* 2009) and also the use of micro-texture for anti-fouling applications (reviewed by Scardino & de Nys 2011). These studies are not included in this synopsis, which focusses on *in situ* conservation actions to promote colonization of biodiversity on marine artificial structures.

Definition: 'Texture' is micro-scale roughness applied to an entire surface that produces depressions and/or elevations ≤1 mm (Strain *et al.* 2018).

See also: *Use environmentally-sensitive material on subtidal artificial structures*; *Create natural rocky reef topography on subtidal artificial structures*; *Create pit habitats (1–50 mm) on subtidal artificial structures*; *Create groove habitats (1–50 mm) on subtidal artificial structures*; *Create small protrusions (1–50 mm) on subtidal artificial structures; Create small ridges or ledges (1–50 mm) on subtidal artificial structures*; *Create groove habitats* and *small protrusions, ridges or ledges (1–50 mm) on subtidal artificial structures*.

Bourget E., DeGuise J. & Daigle G. (1994) Scales of substratum heterogeneity, structural complexity, and the early establishment of a marine epibenthic community. *Journal of Experimental Marine Biology and Ecology*, 181, 31–51.

Carl C., Poole A.J., Sexton B.A., Glenn F.L., Vucko M.J., Williams M.R., Whalan S. & de Nys R. (2012) Enhancing the settlement and attachment strength of pediveligers of *Mytilus galloprovincialis* by changing surface wettability and microtopography. *Biofouling: The Journal of Bioadhesion and Biofilm Research*, 28, 175–186.

Maldonado M. & Uriz M.J. (1998) Microrefuge exploitation by subtidal encrusting sponges: patterns of settlement and post-settlement survival. *Marine Ecology Progress Series*, 174, 141–150.

Miller M.W. & Barimo J. (2001) Assessment of juvenile coral populations at two reef restoration sites in the Florida Keys National Marine Sanctuary: indicators of success? *Bulletin of Marine Science*, 69, 395–405.

Neo M.L., Todd P.A., Teo S.L.-M. & Chou L.M. (2009) Can artificial substrates enriched with crustose coralline algae enhance larval settlement and recruitment in the fluted giant clam (*Tridacna squamosa*)? *Hydrobiologia*, 625, 83–90.

Scardino A.J. & de Nys R. (2011) Mini review: biomimetic models and bioinspired surfaces for fouling control. *Biofouling: The Journal of Bioadhesion and Biofilm Research*, 27, 73–86.

Sedano F., Navarro-Barranco C., Guerra-García J.M. & Espinosa F. (2020) Understanding the effects of coastal defence structures on marine biota: the role of substrate composition and roughness in structuring sessile, macro- and meiofaunal communities. *Marine Pollution Bulletin*, 157, 111334.

Sempere-Valverde J., Ostalé-Valriberas E., Farfán G.M. & Espinosa F. (2018) Substratum type affects recruitment and development of marine assemblages over artificial substrata: a case study in the Alboran Sea. *Estuarine, Coastal and Shelf Science*, 204, 56–65.

Strain E.M.A., Olabarria C., Mayer-Pinto M., Cumbo V., Morris R.L., Bugnot A.B., Dafforn K.A., Heery E., Firth L.B., Brooks P.R. & Bishop M.J. (2018) Eco-engineering urban infrastructure for marine and coastal biodiversity: which interventions have the greatest ecological benefit? *Journal of Applied Ecology*, 55, 426–441.

Wahl M. & Hoppe K. (2002) Interactions between substratum rugosity, colonization density and periwinkle grazing efficiency. *Marine Ecology Progress Series*, 225, 239–249.

A replicated, randomized, controlled study in 2005 on three subtidal rocky reefs on open coastlines in the Adriatic Sea and the Ionian Sea, Italy (1) found that settlement plates with and without textured surfaces supported similar macroalgae and non-mobile invertebrate species richness, live cover and community composition, while abundances varied depending on the species group and site. After nine months, there was no clear difference in the macroalgae and non-mobile invertebrate community composition, species richness or live cover on plates with and without textured surfaces (data reported as statistical model results). Non-mobile invertebrates were more abundant on plates with texture (<1–6% cover) than without (<1–2%) but the difference was only significant at one of six sites. Macroalgal abundances varied by species group and site (see paper for results). Limestone, sandstone, granite and concrete settlement plates (150 × 100 mm) were made with and without textured surfaces. Five of each material-texture combination were randomly arranged, horizontally at 5 m depth in each of two sites on each of three limestone rocky reefs in February 2005. Macroalgae and non-mobile invertebrates on plates were counted in the laboratory over nine months.

A replicated, randomized, controlled study (year not reported) on open coastlines in the Mediterranean Sea and the Gulf of Aqaba, Israel (2) found that upward-facing settlement plates with textured surfaces supported similar macroalgae and non-mobile invertebrate abundance but different community composition to downward-facing surfaces without texture. After 12 months, macroalgae and non-mobile invertebrate live cover was similar on upward-facing settlement plate surfaces with texture (81–100%) and downward-facing surfaces without (80–100%), but the community composition differed (data reported as statistical model results). Concrete settlement plates (150 × 150 mm) were moulded with textured surfaces on one side and flat on the other, using a formliner. Plates were either standard-concrete or one of five patented ECOncrete™ materials. Ten of each material were randomly arranged horizontally with textured surfaces facing upwards on frames at 6 m depth in the Mediterranean Sea and at 10 m in the Gulf of Aqaba (month/year not reported). Macroalgae and non-mobile invertebrates on plates were counted and biomass (dry weight) was recorded in the laboratory over 12 months.

A replicated, controlled study in 2013–2014 on 24 jetty pilings in the Hudson River estuary, USA (3) found that creating textured surfaces on the pilings, along with using environmentally-sensitive material, increased the macroalgae and invertebrate species richness, cover and biomass and altered the community composition on piling surfaces. After 14 months, pilings with textured surfaces and environmentally-sensitive material supported 18 macroalgae and invertebrate species with 90–100% cover, while fibreglass pilings without texture supported nine species with 40–85% cover (data not statistically tested). Biomass was higher on pilings with textured surfaces (0.07 g/cm^2) than without (0.02 g/cm^2) and the community composition differed (data reported as statistical model results). Over 14 months, six species (4 non-mobile invertebrates, 2 mobile invertebrates) recorded on pilings with texture were absent from those without. It is not clear whether these effects were the direct result of creating textured surfaces or using environmentally-sensitive material. Textured surfaces were created on concrete jetty piling encasements using a formliner during maintenance works. Nine textured encasements made from patented ECOncrete™ material and three untextured fibreglass encasements were attached around pilings in each of two sites along a jetty in June 2013 (depth not reported). Macroalgae and invertebrates were counted on and around pilings and biomass was measured (dry weight) in the laboratory over 14 months.

(1) Guarnieri G., Terlizzi A., Bevilacqua S. & Fraschetti S. (2009) Local vs regional effects of substratum on early colonization stages of sessile assemblages. *Biofouling: The Journal of Bioadhesion and Biofilm Research*, 25, 593–604.
(2) Perkol-Finkel S. & Sella I. (2014) Ecologically-active concrete for coastal and marine infrastructure: innovative matrices and designs. Pages 1139–1149 in: W. Allsop & K. Burgess (eds.) *From Sea to Shore – Meeting the Challenges of the Sea (Coasts, Marine Structures and Breakwaters 2013)*. ICE publishing.

(3) Perkol-Finkel S. & Sella I. (2016) Blue is the new green – harnessing urban coastal infrastructure for ecological enhancement. Pages 139–149 in: A. Baptiste (ed.) *Coastal Management: Changing Coast, Changing Climate, Changing Minds*. ICE publishing.

3.3. Create natural rocky reef topography on subtidal artificial structures

- **One study** examined the effects of creating natural rocky reef topography on subtidal artificial structures on the biodiversity of those structures. The study was on an open coastline in Italy[1].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (1 STUDY)

- **Algal abundance (1 study):** One replicated, randomized, controlled study in Italy[1] found that creating natural rocky reef topography on subtidal artificial structures did not increase the abundance of juvenile canopy macroalgae that settled onto structure surfaces, regardless of the topography depth.

BEHAVIOUR (0 STUDIES)

Background

Topography influences the settlement and survival of marine organisms on subtidal rocky substrates. Variation in topography generates variation in the physical environment and plays an important role in sustaining biodiversity and ecological functioning (Levin 1974). On rocky reefs, many habitat features that offer refuge from physical stressors and predation, such as bumps, crevices and holes, are generated as a function of substrate topography and geomorphology. The full fingerprint of natural rocky reef topography encompasses a variety of habitat features of different scales interacting within a mosaic.

Marine artificial structures often have much lower topographic variability than natural rocky reefs, which is thought to be a key reason for their reduced biodiversity (Wilhelmsson & Malm 2008). Natural rocky reef topography can be created on subtidal artificial structures by moulding or casting material during construction or retrospectively (see Evans *et al.* 2021).

Definition: 'Natural rocky reef topography' refers to the full fingerprint of substrate topography found in natural rocky habitats.

See also: *Create textured surfaces (≤1 mm) on subtidal artificial structures*; *Create pit habitats (1–50 mm) on subtidal artificial structures*; *Create hole habitats (>50 mm) on subtidal artificial structures*; *Create groove habitats (1–50 mm) on subtidal artificial structures*; *Create crevice habitats (>50 mm) on subtidal artificial structures*; *Create small protrusions (1–50 mm) on subtidal artificial structures*; *Create large protrusions (>50 mm) on subtidal artificial structures; Create small ridges or ledges (1–50 mm) on subtidal* *artificial structures*; *Create large ridges or ledges (>50 mm) on subtidal artificial structures*; *Create groove habitats* and *small protrusions, ridges or ledges (1–50 mm) on subtidal artificial structures*.

Evans A.J., Lawrence P.J., Natanzi A.S., Moore P.J., Davies A.J., Crowe T.P., McNally C., Thompson B., Dozier A.E. & Brooks P.R. (2021) Replicating natural topography on marine artificial structures – a novel approach to eco-engineering. *Ecological Engineering*, 160, 106144.

Levin S.A. (1974) Dispersion and population interactions. *American Society of Naturalists*, 108, 207–228.

Wilhelmsson D. & Malm T. (2008) Fouling assemblages on offshore wind power plants and adjacent substrata. *Estuarine, Coastal and Shelf Science*, 79, 459–466.

A replicated, randomized, controlled study in 2009 on a subtidal rocky reef on open coastline in the Adriatic Sea, Italy (1) found that creating natural rocky reef topography on settlement plates did not increase the abundance of juvenile canopy algae *Cystoseira barbata* that settled onto plates. After five months, there was no significant difference in the average abundance of juveniles on settlement plates with natural rocky reef topography (deep topography: 23/plate; shallow: 28/plate) and plates without (19/plate). Clay settlement plates (100 × 100 mm) were made with and without natural rocky reef topography imprinted on their surfaces using pieces of natural rock as the clay set. Six plates with each of deep (imprinted 5 mm deep) and shallow (1–2 mm) topography and six plates without were randomly arranged horizontally at 3 m depth on a rocky reef with existing adult canopy algae in March 2009. Juvenile canopy algae on plates were counted after five months.

(1) Perkol-Finkel S., Ferrario F., Nicotera V. & Airoldi L. (2012) Conservation challenges in urban seascapes: promoting the growth of threatened species on coastal infrastructures. *Journal of Applied Ecology*, 49, 1457–1466.

3.4. Create pit habitats (1–50 mm) on subtidal artificial structures

- **One study** examined the effects of creating pit habitats on subtidal artificial structures on the biodiversity of those structures. The study was on an open coastline in northern Israel[1].

COMMUNITY RESPONSE (1 STUDY)

- **Overall community composition (1 study):** One replicated, controlled study in Israel[1] found that pit habitats created on a subtidal artificial structure, along with holes, grooves and environmentally-sensitive material, altered the combined macroalgae and invertebrate community composition on structure surfaces. They also supported macroalgae, non-mobile invertebrate and fish species that were absent from a similar structure without the added habitat features.
- **Overall richness/diversity (1 study):** One replicated, controlled study in Israel[1] found that creating pit habitats on a subtidal artificial structure, along with holes, grooves and environmentally-sensitive material, increased the combined macroalgae and invertebrate species diversity on structure surfaces.

POPULATION RESPONSE (1 STUDY)

- **Algal abundance (1 study):** One replicated, controlled study in Israel[1] reported that creating pit habitats on a subtidal artificial structure, along with holes, grooves and environmentally-sensitive material, had mixed effects on macroalgal abundances on structure surfaces, depending on the species group.
- **Invertebrate abundance (1 study):** One replicated, controlled study in Israel[1] reported that creating pit habitats on a subtidal artificial structure, along with holes, grooves and environmentally-sensitive material, had mixed effects on invertebrate abundances on structure surfaces, depending on the species group.
- **Fish abundance (1 study):** One replicated, controlled study in Israel[1] reported that creating pit habitats on a subtidal artificial structure, along with holes, grooves and environmentally-sensitive material, had mixed effects on fish abundances on and around structure surfaces, depending on the species group.

BEHAVIOUR (0 STUDIES)

Background

Pit habitats provide organisms refuge from predation in subtidal rocky habitats (Nelson & Vance 1979). Some species preferentially settle into them (Nozawa *et al.* 2011). The size and density of pits is likely to affect the size, abundance and variety of organisms that can use them. Small pits can provide refuge for small-bodied organisms but may exclude larger organisms, limit their growth and get rapidly filled-up (Firth *et al.* 2020). Large pits can be used by larger-bodied organisms but may not provide sufficient refuge from predators for smaller organisms.

Pits are sometimes present on boulders used in marine artificial structures as a result of quarrying processes (Hall *et al.* 2018), and can form on other structures through erosion. However, these are often filled or repaired during maintenance works (Moreira *et al.* 2007) and are absent from many structures. Pit habitats can be created on subtidal artificial structures by adding or removing material, either during construction or retrospectively.

There is a body of literature investigating the effects of creating pit habitats on artificial substrates for coral rearing and gardening (e.g. Nozawa *et al.* 2011; Okamoto *et al.* 2010). These studies are not included in this synopsis, which focusses on *in situ* conservation actions to enhance the biodiversity of structures that are engineered to fulfil a primary function other than providing artificial habitats.

Definition: 'Pit habitats' are depressions with a length to width ratio ≤3:1 and depth 1–50 mm (Strain *et al.* 2018).

See also: *Create textured surfaces (≤1 mm) on subtidal artificial structures*; *Create natural rocky reef topography on subtidal artificial structures*; *Create hole habitats (>50 mm) on* *subtidal artificial structures*; *Create groove habitats (1–50 mm) on subtidal artificial structures*; *Create crevice habitats (>50 mm) on subtidal artificial structures*; *Create groove habitats* and *small protrusions, ridges or ledges (1–50 mm) on subtidal artificial structures*.

Firth L.B., Airoldi L., Bulleri F., Challinor S., Chee S.-Y., Evans A.J., Hanley M.E., Knights A.M., O'Shaughnessy K., Thompson R.C. & Hawkins S.J. (2020) Greening of grey infrastructure should not be used as a Trojan horse to facilitate coastal development. *Journal of Applied Ecology*, 57, 1762–1768.

Hall A.E., Herbert R.J.H., Britton J.R. & Hull S.L. (2018) Ecological enhancement techniques to improve habitat heterogeneity on coastal defence structures. *Estuarine, Coastal and Shelf Science*, 210, 68–78.

Moreira J., Chapman M.G. & Underwood A.J. (2007) Maintenance of chitons on seawalls using crevices on sandstone blocks as habitat in Sydney Harbour, Australia. *Journal of Experimental Marine Biology and Ecology*, 347, 134–143.

Nelson B.V. & Vance R.R. (1979) Diel foraging patterns of the sea urchin *Centrostephanus coronatus* as a predator avoidance strategy. *Marine Biology*, 51, 251–258.

Nozawa Y., Tanaka K. & Reimer J.D. (2011) Reconsideration of the surface structure of settlement plates used in coral recruitment studies. *Zoological Studies*, 50, 53–60.

Okamoto M., Yap M., Roeroe A.K., Nojima S., Oyamada K., Fujiwara S. & Iwata I. (2010) *In situ* growth and mortality of juvenile *Acropora* over 2 years following mass spawning in Sekisei Lagoon, Okinawa (24°N). *Fisheries Science*, 76, 343–353.

Strain E.M.A., Olabarria C., Mayer-Pinto M., Cumbo V., Morris R.L., Bugnot A.B., Dafforn K.A., Heery E., Firth L.B., Brooks P.R. & Bishop M.J. (2018) Eco-engineering urban infrastructure for marine and coastal biodiversity: which interventions have the greatest ecological benefit? *Journal of Applied Ecology*, 55, 426–441.

A replicated, controlled study in 2012–2014 on two subtidal breakwaters on open coastline in the Mediterranean Sea, Israel (1) found that pit habitats created on breakwater blocks, along with holes, grooves and environmentally-sensitive material, supported different macroalgae and invertebrate community composition with higher species diversity than standard-concrete blocks without added habitats, while macroalgae, invertebrate and fish abundances varied depending on the species group. After 24 months, the macroalgae and invertebrate species diversity was higher on blocks with added habitats than without (data reported as Shannon index) and the community composition differed (data reported as statistical model results). Thirty species (7 mobile invertebrates, 14 non-mobile invertebrates, 9 fishes) recorded on and around blocks with added habitats were absent from blocks without. Species abundances varied on blocks with and without added habitats depending on the species group (see paper for results). It is not clear whether these effects were the direct result of creating pits, holes, grooves, or using environmentally-sensitive material. Pit habitats were created on breakwater blocks (1 × 1 × 1 m) using a formliner. Each block had multiple round pits (diameter: 10 mm; depth: 5 mm; T. Hadary *pers. comms.*) amongst multiple holes and grooves (number/spacing not reported). Five blocks of each of three patented ECOncrete™ materials (lower pH and different cement/additives to standard-concrete) were placed at 5–7 m depth on a concrete-block breakwater during construction in July 2012. Five standard-concrete blocks (1.7 × 1.7 × 1.7 m) without added habitats were placed on a similar breakwater 80 m away. Macroalgae and invertebrates on blocks, and fishes on and around blocks, were counted over 24 months.

(1) Sella I. & Perkol-Finkel S. (2015) Blue is the new green – ecological enhancement of concrete based coastal and marine infrastructure. *Ecological Engineering*, 84, 260–272.

3.5. Create hole habitats (>50 mm) on subtidal artificial structures

- **Three studies** examined the effects of creating hole habitats on subtidal artificial structures on the biodiversity of those structures. One study was on an open coastline in northern Israel[1], one was in a marina in northern Israel[2], and one was off the west coast of Sweden[3].

COMMUNITY RESPONSE (3 STUDIES)

- **Overall community composition (3 studies):** Three replicated, controlled studies (including one randomized, paired sites, before-and-after study) in Israel[1,2] and off Sweden[3] found that creating hole habitats on subtidal artificial structures, along with grooves[1,2], environmentally-sensitive material[1,2] and pits[1] or small ledges[2] in two studies, altered the combined macroalgae and invertebrate[1,2] or mobile invertebrate and fish[3] community composition on and around structures. They also supported mobile invertebrate[1,3], non-mobile invertebrate[1,2] and/or fish[1] species that were absent from structure surfaces without added habitat features.
- **Overall richness/diversity (3 studies):** Two of three replicated, controlled studies (including one randomized, paired sites, before-and-after study) in Israel[1,2] and off Sweden[3] found that creating hole habitats on subtidal artificial structures, along with grooves[1,2], environmentally-sensitive material[1,2] and pits[1] or small ledges[2], increased the combined macroalgae and invertebrate species richness[2] and/or diversity[1,2] on and around structures. One[3] found that creating holes did not increase the combined mobile invertebrate and fish species richness or diversity.

POPULATION RESPONSE (2 STUDIES)

- **Overall abundance (1 study):** One replicated, controlled study off Sweden[3] reported that creating hole habitats on subtidal artificial structures did not increase the combined mobile invertebrate and fish abundance on and around structures[3].
- **Algal abundance (1 study):** One replicated, controlled study in Israel[1] reported that creating hole habitats on a subtidal artificial structure, along with pits, grooves and environmentally-sensitive material, had mixed effects on macroalgal abundances on structure surfaces, depending on the species group.
- **Invertebrate abundance (2 studies):** One of two replicated, controlled studies in Israel[1] and off Sweden[3] found that creating hole habitats on subtidal artificial structures increased the abundance of brown crabs on and around structures, but not other mobile invertebrates. One[1] reported that creating holes, along with pits grooves and environmentally-sensitive material, had mixed effects on invertebrate abundances, depending on the species group.
- **Fish abundance (2 studies):** One of two replicated, controlled studies in Israel[1] and off Sweden[3] found that creating hole habitats on subtidal artificial structures did not increase fish species abundances on and around structures. One[1] reported that creating holes, along with pits grooves and environmentally-sensitive material, had mixed effects on fish abundances, depending on the species group.

BEHAVIOUR (0 STUDIES)

Background

Hole habitats provide organisms refuge from predation in subtidal rocky habitats (Nelson & Vance 1979). The size and density of holes is likely to affect the size, abundance and variety of organisms that can use them. Small holes can provide refuge for small-bodied organisms but may exclude larger organisms, limit their growth and get rapidly filled-up (Firth *et al.* 2020). Large holes can be used by larger-bodied organisms but may not provide sufficient refuge from predators for smaller organisms. By default, holes contain shaded surfaces, which can be associated with the presence of non-native species (Dafforn 2017).

Holes are sometimes present on boulders used in marine artificial structures as a result of quarrying processes or engineering tests (Firth *et al*. 2014). However, these are sparse when present and are normally absent from other types of structures. Holes sometimes form on artificial structures through erosion, but are often filled or repaired during maintenance works (Moreira *et al.* 2007). Hole habitats can be created on subtidal artificial structures by adding or removing material, either during construction or retrospectively.

There is a body of literature investigating the effects of creating hole habitats on artificial reefs (reviewed by Morris *et al.* 2018). These studies are not included in this synopsis, which focusses on *in situ* conservation actions to enhance the biodiversity of structures that are engineered to fulfil a primary function other than providing artificial habitats.

Definition: 'Hole habitats' are depressions with a length to width ratio ≤3:1 and depth >50 mm (Strain *et al.* 2018).

See also: *Create natural rocky reef topography on subtidal artificial structures*; *Create pit habitats (1–50 mm) on subtidal artificial structures*; *Create groove habitats (1–50 mm) on subtidal artificial structures*; *Create crevice habitats (>50 mm) on subtidal artificial structures*; *Create small adjoining cavities or 'swimthrough' habitats (≤100 mm) on subtidal artificial structures*; *Create large adjoining cavities or 'swimthrough' habitats (>100 mm) on subtidal artificial structures*; *Create groove habitats* and *small protrusions, ridges or ledges (1–50 mm) on subtidal artificial structures*.

Dafforn K.A. (2017) Eco-engineering and management strategies for marine infrastructures to reduce establishment and dispersal of non-indigenous species. *Management of Biological Invasions*, 8, 153–161.

Firth L.B., Airoldi L., Bulleri F., Challinor S., Chee S.-Y., Evans A.J., Hanley M.E., Knights A.M., O'Shaughnessy K., Thompson R.C. & Hawkins S.J. (2020) Greening of grey infrastructure should not be used as a Trojan horse to facilitate coastal development. *Journal of Applied Ecology*, 57, 1762–1768.

Firth L.B., Thompson R.C., Bohn K., Abbiati M., Airoldi L., Bouma T.J., Bozzeda F., Ceccherelli V.U., Colangelo M.A., Evans A., Ferrario F., Hanley M.E., Hinz H., Hoggart S.P.G., Jackson J.E., Moore P., Morgan E.H., Perkol-Finkel S., Skov M.W., Strain E.M., van Belzen J. & Hawkins S.J. (2014) Between a rock and a hard place: environmental and engineering considerations when designing coastal defence structures. *Coastal Engineering*, 87, 122–135.

Moreira J., Chapman M.G. & Underwood A.J. (2007) Maintenance of chitons on seawalls using crevices on sandstone blocks as habitat in Sydney Harbour, Australia. *Journal of Experimental Marine Biology and Ecology*, 347, 134–143.

Morris R.L., Porter A.G., Figueira W.F., Coleman R.A., Fobert E.K. & Ferrari R. (2018) Fish-smart seawalls: a decision tool for adaptive management of marine infrastructure. *Frontiers in Ecology and the Environment*, 16, 278–287.

Nelson B.V. & Vance R.R. (1979) Diel foraging patterns of the sea urchin *Centrostephanus coronatus* as a predator avoidance strategy. *Marine Biology*, 51, 251–258.

Strain E.M.A., Olabarria C., Mayer-Pinto M., Cumbo V., Morris R.L., Bugnot A.B., Dafforn K.A., Heery E., Firth L.B., Brooks P.R. & Bishop M.J. (2018) Eco-engineering urban infrastructure for marine and coastal biodiversity: which interventions have the greatest ecological benefit? *Journal of Applied Ecology*, 55, 426–441.

A replicated, controlled study in 2012–2014 on two subtidal breakwaters on open coastline in the Mediterranean Sea, Israel (1) found that hole habitats created on breakwater blocks, along with pits, grooves and environmentally-sensitive material, supported different macroalgae and invertebrate community composition with higher species diversity than standard-concrete blocks without added habitats, while macroalgae, invertebrate and fish abundances varied depending on the species group. After 24 months, the macroalgae and invertebrate species diversity was higher on blocks with added habitats than without (data reported as Shannon index) and the community composition differed (data reported as statistical model results). Thirty species (7 mobile invertebrates, 14 non-mobile invertebrates, 9 fishes) recorded on and around blocks with added habitats were absent from blocks without. Species abundances varied on blocks with and without added habitats depending on the species group (see paper for results). It is not clear whether these effects were the direct result of creating holes, pits, grooves, or using environmentally-sensitive material. Hole habitats were created on breakwater blocks (1 × 1 × 1 m) using a formliner. Each block had multiple cube-shaped (60 × 60 × 60 mm), cylindrical (diameter: 30 mm; depth: 100 mm) and hemispherical (diameter: 150 mm; depth: 100 mm) holes (T. Hadary *pers. comms.*) amongst multiple pits and grooves (number/spacing not reported). Five blocks of each of three patented ECOncrete™ materials (lower pH and different cement/additives to standard-concrete) were placed at 5–7 m depth on a concrete-block breakwater during construction in July 2012. Five standard-concrete blocks (1.7 × 1.7 × 1.7 m) without added habitats were placed on a similar breakwater 80 m away. Macroalgae and invertebrates on blocks, and fishes on and around blocks, were counted over 24 months.

A replicated, randomized, paired sites, controlled, before-and-after study in 2014–2016 on a subtidal seawall in a marina in the Mediterranean Sea, Israel (2) found that hole habitats created on seawall panels, along with grooves, small ledges and environmentally-sensitive material, supported higher macroalgae and invertebrate species diversity and richness and different community composition compared with standard-concrete seawall surfaces without added habitats. After 22 months, macroalgae and invertebrate species diversity (data reported as Shannon index) and richness was higher on panels with added habitats (9 species/quadrat) than on seawall surfaces without (5/quadrat), and compared with seawall surfaces before habitats were added (1/quadrat). Community composition differed between panels with added habitats and seawall surfaces without (data reported as statistical model results). Two non-mobile invertebrate species groups recorded on panels were absent from surfaces without. It is not clear whether these effects were the direct result of creating holes, grooves, ledges, or using environmentally-sensitive material. Hole habitats were created on seawall panels (height: 1.5 m; width: 0.9 m; thickness: 130 mm) using a formliner. Each panel had six cylindrical holes (diameter: 30 mm; depth: 120 mm; ≥300 mm apart) amongst multiple grooves and small ledges. Panels were made from patented ECOncrete™ material. Four panels were attached to a vertical concrete seawall in November 2014. The bottom 1.2 m were subtidal. Panels were compared with standard-concrete seawall surfaces cleared of organisms (height: 1.2 m; width: 0.9 m) adjacent to each panel. Macroalgae and invertebrates were counted in one 300 × 300 mm randomly-placed quadrat on each panel and seawall surface over 22 months.

A replicated, controlled study in 2007–2019 on 21 subtidal wave buoy foundations in the North Sea, off the coast of Sweden (3) found that creating hole habitats on foundations did not increase the mobile invertebrate and fish species diversity, richness or overall abundance, but did alter their community composition and increase brown crab *Cancer pagurus* abundances. After 12 years, the mobile invertebrate and fish species diversity (data reported as Shannon and Evenness indices), richness and abundance were similar on and around foundations with holes (10 species/foundation, 51 individuals/foundation) and those without (9 species/foundation, 33 individuals/foundation). The community composition, however, differed (data reported as statistical model results). Three mobile invertebrate species recorded on and around foundations with holes were absent from those without. Brown crabs were more abundant on and around foundations with holes (11/foundation) than without (4/foundation), while the abundances of 47 other fish and mobile invertebrates were similar for both (see paper for results). Hole habitats were created in April 2007 by drilling into the vertical sides of concrete foundations (diameter: 3 m; height: 1 m). There were 26 evenly-spaced cuboidal holes/foundation (width: 120 mm; height: 150 mm; depth: 300 mm): 13 at seabed level and 13 at 0.5 m above the seabed. Eleven foundations with holes and 10 without were placed on sandy seabed at 25 m depth. Fishes and mobile invertebrates were counted on and around (<1 m radius) foundations over 12 years. Holes at seabed level had been buried by sediment and no longer provided habitats.

(1) Sella I. & Perkol-Finkel S. (2015) Blue is the new green – ecological enhancement of concrete based coastal and marine infrastructure. *Ecological Engineering*, 84, 260–272.

(2) Perkol-Finkel S., Hadary T., Rella A., Shirazi R. & Sella I. (2018) Seascape architecture – incorporating ecological considerations in design of coastal and marine infrastructure. *Ecological Engineering*, 120, 645–654.

(3) Bender A., Langhamer O. & Sundberg J. (2020) Colonisation of wave power foundations by mobile mega- and macrofauna – a 12 year study. *Marine Environmental Research*, 161, 105053.

3.6. Create groove habitats (1–50 mm) on subtidal artificial structures

- **Two studies** examined the effects of creating groove habitats on subtidal artificial structures on the biodiversity of those structures. Both studies were on open coastlines in Japan[1] and northern Israel[2].

COMMUNITY RESPONSE (1 STUDY)

- **Overall community composition (1 study):** One replicated, controlled study in Israel[2] found that groove habitats created on a subtidal artificial structure, along with holes, pits and environmentally-sensitive material, altered the combined macroalgae and invertebrate community composition on structure surfaces. They also supported macroalgae, non-mobile invertebrate and fish species that were absent from a similar structure without added habitat features.
- **Overall richness/diversity (1 study):** One replicated, controlled study in Israel[2] found that creating groove habitats on a subtidal artificial structure, along with holes, pits and environmentally-sensitive material, increased the combined macroalgae and invertebrate species diversity on structure surfaces.

POPULATION RESPONSE (2 STUDIES)

- **Algal abundance (2 studies):** Two controlled studies (including one replicated study) in Japan[1] and Israel[2] reported that creating groove habitats on subtidal artificial structures, along with holes, pits and environmentally-sensitive material in one[2], had mixed effects on macroalgal abundances on structure surfaces, depending on the species group.
- **Invertebrate abundance (1 study):** One replicated, controlled study in Israel[2] reported that creating groove habitats on a subtidal artificial structure, along with holes, pits and environmentally-sensitive material, had mixed effects on invertebrate abundances on structure surfaces, depending on the species group.
- **Fish abundance (1 study):** One replicated, controlled study in Israel[2] reported that creating groove habitats on a subtidal artificial structure, along with holes, pits and environmentally-sensitive material, had mixed effects on fish abundances on and around structure surfaces, depending on the species group.

BEHAVIOUR (0 STUDIES)

Background

Groove habitats provide organisms refuge from predation in subtidal rocky habitats (Nelson & Vance 1979). Some species preferentially settle into them (Bourget *et al.* 1994). The size and density of grooves is likely to affect the size, abundance and variety of organisms that can use them. Small grooves can provide refuge for small-bodied organisms but may exclude larger organisms, limit their growth and get rapidly filled-up (Firth *et al.* 2020). Large grooves can be used by larger-bodied organisms but may not provide sufficient refuge from predators for smaller organisms.

Grooves are sometimes present on artificial structures such as cable mattresses (Lacey & Hayes 2020) or quarried boulders (MacArthur *et al.* 2020). They can also form on structures through erosion, but will often be filled or repaired during maintenance works (Moreira *et al.* 2007), and are absent from many structures. Groove habitats can be created on subtidal artificial structures by adding or removing material, either during construction or retrospectively.

There is a body of literature investigating the effects of creating groove habitats on artificial reefs and on substrates for coral rearing or gardening (e.g. Douke *et al.* 1998; Rani *et al.* 2015). These studies are not included in this synopsis, which focusses on *in situ* conservation actions to enhance the biodiversity of structures that are engineered to fulfil a primary function other than providing artificial habitats.

Definition: 'Groove habitats' are depressions with a length to width ratio >3:1 and depth 1–50 mm (modified from "Crevices" in Strain *et al.* 2018).

See also: *Create textured surfaces (≤1 mm) on subtidal artificial structures*; *Create natural rocky reef topography on subtidal artificial structures*; *Create pit habitats (1–50 mm) on subtidal artificial structures*; *Create hole habitats (>50 mm) on subtidal artificial structures*; *Create crevice habitats (>50 mm) on subtidal artificial structures*; *Create groove habitats* and *small protrusions, ridges or ledges (1–50 mm) on subtidal artificial structures*.

Bourget E., DeGuise J. & Daigle G. (1994) Scales of substratum heterogeneity, structural complexity, and the early establishment of a marine epibenthic community. *Journal of Experimental Marine Biology and Ecology*, 181, 31–51.

Douke A., Munekiyo M., Tsuji S. & Itani M. (1998) Effect of artificial reef for catch topshell, *Batillus cornutus*. *Fisheries Engineering* (Japan), accessed from *The Agriculture, Forestry and Fisheries Research Information Technology Center*, 35, 145–152.

Firth L.B., Airoldi L., Bulleri F., Challinor S., Chee S.-Y., Evans A.J., Hanley M.E., Knights A.M., O'Shaughnessy K., Thompson R.C. & Hawkins S.J. (2020) Greening of grey infrastructure should not be used as a Trojan horse to facilitate coastal development. *Journal of Applied Ecology*, 57, 1762–1768.

Lacey N.C. & Hayes P. (2020) Epifauna associated with subsea pipelines in the North Sea. *ICES Journal of Marine Science*, 77, 1137–1147.

MacArthur M., Naylor L.A., Hansom J.D. & Burrows M.T. (2020) Ecological enhancement of coastal engineering structures: passive enhancement techniques. *Science of the Total Environment*, 740, 139981.

Moreira J., Chapman M.G. & Underwood A.J. (2007) Maintenance of chitons on seawalls using crevices on sandstone blocks as habitat in Sydney Harbour, Australia. *Journal of Experimental Marine Biology and Ecology*, 347, 134–143.

Nelson B.V. & Vance R.R. (1979) Diel foraging patterns of the sea urchin *Centrostephanus coronatus* as a predator avoidance strategy. *Marine Biology*, 51, 251–258.

Rani M.H., Saad S., Khodzari M.F.A., Ramli R. & Yusof M.H. (2015) Scleractinian coral recruitment density in coastal water of Balok, Pahang, Malaysia. *Jurnal Teknologi (Sciences & Engineering)*, 77, 13–18.

Strain E.M.A., Olabarria C., Mayer-Pinto M., Cumbo V., Morris R.L., Bugnot A.B., Dafforn K.A., Heery E., Firth L.B., Brooks P.R. & Bishop M.J. (2018) Eco-engineering urban infrastructure for marine and coastal biodiversity: which interventions have the greatest ecological benefit? *Journal of Applied Ecology*, 55, 426–441.

A controlled study in 1985–1989 on a subtidal breakwater block on open coastline in Toyama Bay, Japan (1) reported that groove habitats created on the block supported more kelp *Ecklonia stolonifera* but similar abundances of canopy algae *Sargassum* spp. compared with a block surface without grooves. Data were not statistically tested. After 42 months, there were 55 kelp individuals on the surface with large grooves (wet weight: 0.93 kg), 32 on the surface with small grooves (0.48 kg) and 20 on the surface without grooves (0.31 kg). Three canopy algae species had similar abundances and weights on the surface with large grooves (5–10 individuals, all 0.01 kg), small grooves (2–19 individuals, 0.01–0.04 kg) and without grooves (3–18 individuals, 0.05–0.17 kg). Groove habitats were created on a concrete breakwater block (2.3 × 2.3 × 0.8 m). There was one array of five large grooves (length: 644 mm; width: 46 mm; depth: 23 mm) and one of nine small grooves (length: 644 mm; width: 3 mm; depth not reported), evenly-spaced on 644 × 529 mm horizontal surfaces. One adjacent surface had no grooves. Small grooves were created by scraping using a nail (method for large grooves not reported). The block was placed on sandy seabed at 9 m depth in November 1985. Macroalgae on surfaces with and without grooves were counted and weighed (wet weight) after 42 months.

A replicated, controlled study in 2012–2014 on two subtidal breakwaters on open coastline in the Mediterranean Sea, Israel (2) found that groove habitats created on breakwater blocks, along with holes, pits and environmentally-sensitive material, supported different macroalgae and invertebrate community composition with higher species diversity than standard-concrete blocks without added habitats, while macroalgae, invertebrate and fish abundances varied depending on the species group. After 24 months, the macroalgae and invertebrate species diversity was higher on blocks with added habitats than without (data reported as Shannon index) and the community composition differed (data reported as statistical model results). Thirty species (7 mobile invertebrates, 14 non-mobile invertebrates, 9 fishes) recorded on and around blocks with added habitats were absent from blocks without. Species abundances varied on blocks with and without added habitats depending on the species group (see paper for results). It is not clear whether these effects were the direct result of creating grooves, holes, pits, or using environmentally-sensitive material. Groove habitats were created on breakwater blocks (1 × 1 × 1 m) using a formliner. Each block had multiple irregular grooves (length: 100–600 mm; width: 5–15 mm; depth: 10 mm; T. Hadary *pers. comms.*) amongst multiple holes and pits (number/spacing not reported). Five blocks of each of three patented ECOncrete™ materials (lower pH and different cement/additives to standard-concrete) were placed at 5–7 m depth on a concrete-block breakwater during construction in July 2012. Five standard-concrete blocks (1.7 × 1.7 × 1.7 m) without added habitats were placed on a similar breakwater 80 m away. Macroalgae and invertebrates on blocks, and fishes on and around blocks, were counted over 24 months.

(1) Watanuki A. & Yamamoto H. (1990) Settlement of seaweeds on coastal structures. *Hydrobiologia*, 204, 275–280.

(2) Sella I. & Perkol-Finkel S. (2015) Blue is the new green – ecological enhancement of concrete based coastal and marine infrastructure. *Ecological Engineering*, 84, 260–272.

3.7. Create crevice habitats (>50 mm) on subtidal artificial structures

- We found no studies that evaluated the effects of creating crevice habitats on subtidal artificial structures on the biodiversity of those structures.

This means we did not find any studies that directly evaluated this intervention during our literature searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Background

Crevice habitats provide organisms refuge from predation in subtidal rocky habitats (Nelson & Vance 1979). The size and density of crevices is likely to affect the size, abundance and variety of organisms that can use them. Small crevices can provide refuge for small-bodied organisms but may exclude larger organisms, limit their growth and get rapidly filled-up (Firth *et al.* 2020). Large crevices can be used by larger-bodied organisms but may not provide sufficient refuge from predators for smaller organisms. By default, crevices contain shaded surfaces, which can be associated with the presence of non-native species (Dafforn 2017).

Crevices are sometimes present on marine artificial structures such as cable mattresses (Lacey & Hayes 2020) but are often absent from other types of structures. Crevices sometimes form on structures through erosion, but are often filled or repaired during maintenance works (Moreira *et al.* 2007). Crevice habitats can be created on subtidal artificial structures by adding or removing material, either during construction or retrospectively.

There are bodies of literature investigating the effects of creating crevice habitats on artificial reefs (e.g. Briones-Fourzán *et al.* 2007) and the use of crevice units to sample small/cryptic fauna (e.g. Baronia & Bucher 2008). These studies are not included in this synopsis, which focusses on *in situ* conservation actions to enhance the biodiversity of structures that are engineered to fulfil a primary function other than providing artificial habitats.

Definition: 'Crevice habitats' are depressions with a length to width ratio >3:1 and depth >50 mm (modified from "Crevices" in Strain *et al.* 2018).

See also: *Create natural rock reef topography on subtidal artificial structures*; *Create pit habitats (1–50 mm) on subtidal artificial structures*; *Create hole habitats (>50 mm) on subtidal artificial structures*; *Create groove habitats (1–50 mm) on subtidal artificial structures*; *Create small adjoining cavities or 'swimthrough' habitats (≤100 mm) on subtidal artificial structures*; *Create large adjoining cavities or 'swimthrough' habitats (>100 mm) on subtidal artificial structures*; *Create groove habitats* and *small protrusions, ridges or ledges (1–50 mm) on subtidal artificial structures*.

Baronio M. de A. & Bucher D.J. (2008) Artificial crevice habitats to assess the biodiversity of vagile macro-cryptofauna of subtidal rocky reefs. *Marine and Freshwater Research*, 59, 661–670.
Briones-Fourzán P., Lozano-Álvarez E., Negrete-Soto F. & Barradads-Ortiz C. (2007) Enhancement of juvenile Caribbean spiny lobsters: an evaluation of changes in multiple response variables with the addition of large artificial shelters. *Population Ecology*, 151, 401–416.
Dafforn K.A. (2017) Eco-engineering and management strategies for marine infrastructures to reduce establishment and dispersal of non-indigenous species. *Management of Biological Invasions*, 8, 153–161.
Firth L.B., Airoldi L., Bulleri F., Challinor S., Chee S.-Y., Evans A.J., Hanley M.E., Knights A.M., O'Shaughnessy K., Thompson R.C. & Hawkins S.J. (2020) Greening of grey infrastructure should not be used as a Trojan horse to facilitate coastal development. *Journal of Applied Ecology*, 57, 1762–1768.
Lacey N.C. & Hayes P. (2020) Epifauna associated with subsea pipelines in the North Sea. *ICES Journal of Marine Science*, 77, 1137–1147.
Moreira J., Chapman M.G. & Underwood A.J. (2007) Maintenance of chitons on seawalls using crevices on sandstone blocks as habitat in Sydney Harbour, Australia. *Journal of Experimental Marine Biology and Ecology*, 347, 134–143.
Nelson B.V. & Vance R.R. (1979) Diel foraging patterns of the sea urchin *Centrostephanus coronatus* as a predator avoidance strategy. *Marine Biology*, 51, 251–258.
Strain E.M.A., Olabarria C., Mayer-Pinto M., Cumbo V., Morris R.L., Bugnot A.B., Dafforn K.A., Heery E., Firth L.B., Brooks P.R. & Bishop M.J. (2018) Eco-engineering urban infrastructure for marine and coastal biodiversity: which interventions have the greatest ecological benefit? *Journal of Applied Ecology*, 55, 426–441.

3.8. Create small adjoining cavities or 'swimthrough' habitats (≤100 mm) on subtidal artificial structures

- **Four studies** examined the effects of creating small adjoining cavities or 'swimthrough' habitats on subtidal artificial structures on the biodiversity of those structures. Two studies were in marinas in France[2] and Morocco[4], while one was in each of a lagoon in Mayotte[1] and a port in France[3].

COMMUNITY RESPONSE (1 STUDY)

- **Fish community composition (1 study):** One replicated, paired sites, controlled study in France[3] found that creating small swimthrough habitats on subtidal artificial structures had mixed effects on the juvenile fish community composition on and around structure surfaces, depending on the site and survey month. Swimthrough habitats supported six species that were absent from structure surfaces without swimthroughs.
- **Fish richness/diversity (1 study):** One replicated, paired sites, controlled study in France[3] found that creating small swimthrough habitats on subtidal artificial structures had mixed effects on juvenile fish species richness on and around structure surfaces, depending on the site.

POPULATION RESPONSE (2 STUDIES)

- **Fish abundance (2 studies):** Two replicated, paired sites, controlled studies in France[2,3] found that creating small swimthrough habitats on subtidal artificial structures had mixed effects on juvenile fish abundances on and around structure surfaces, depending on the species[2,3], site[2,3], survey month[2,3] and/or juvenile development stage[2].

BEHAVIOUR (3 STUDIES)

- **Use (3 studies):** One replicated, paired sites, controlled study in France[2] found that creating small swimthrough habitats on subtidal artificial structures had mixed effects on juvenile seabream habitat use on and around structure surfaces, depending on the species and juvenile development stage. Two studies (including one replicated study) in Mayotte[1] and Morocco[4] reported that small swimthrough habitats, along with large swimthroughs and environmentally-sensitive material in one[1], were used by juvenile spiny lobsters, sea firs, adult fish[1] and/or juvenile fish[1,4].

Background

Small adjoining cavities or 'swimthrough' habitats are not well-studied in subtidal rocky habitats. They may form through weathering of softer rocks, amongst loosely-consolidated cobbles/boulders, or within three-dimensional structures created by living organisms. They likely provide organisms refuge from predation, in the same way crevice and hole habitats do (Mercader *et al.* 2019; Nelson & Vance 1979). They could also serve as corridors, connecting adjacent refuge habitats. The size and density of cavities or swimthroughs is likely to affect the size, abundance and variety of organisms that can use them. Small habitats can provide refuge for small-bodied organisms but may exclude larger organisms, limit their growth and get rapidly filled-up (Firth *et al.* 2020). Large habitats can be used by larger-bodied organisms but may not provide sufficient refuge from predators for smaller organisms. By default, cavities and swimthroughs contain shaded surfaces, which can be associated with the presence of non-native species (Dafforn 2017).

Cavities/swimthroughs are sometimes present on marine artificial structures made of consolidated boulders or blocks (Sherrard *et al.* 2016) or gabion baskets (Firth *et al.* 2014), but are absent from many other structures. Small adjoining cavities or 'swimthrough' habitats can be created on subtidal artificial structures by adding or removing material, either during construction or retrospectively.

There is a body of literature investigating the effects of creating swimthrough habitats on artificial reefs (e.g. Brotto *et al.* 2006; Hylkema *et al.* 2020; Noordin Raja Omar *et al.* 1994). These studies are not included in this synopsis, which focusses on *in situ* conservation actions to enhance the biodiversity of structures that are engineered to fulfil a primary function other than providing artificial habitats.

Definition: 'Small adjoining cavities or 'swimthrough' habitats' are adjoining internal cavities sheltered from, but with access to/from, outside the structure. Dimensions can vary but are ≤100 mm in any direction.

See also: *Create hole habitats (>50 mm) on subtidal artificial structures*; *Create crevice habitats (>50 mm) on subtidal artificial structures*; *Create large adjoining cavities or 'swimthrough' habitats (>100 mm) on subtidal artificial structures*.

Brotto D.S., Krohling W., Brum S. & Zalmon I.R. (2006) Usage patterns of an artificial reef by the fish community on the northern coast of Rio de Janeiro – Brazil. *Journal of Coastal Research*, 39, 1276–1280.
Dafforn K.A. (2017) Eco-engineering and management strategies for marine infrastructures to reduce establishment and dispersal of non-indigenous species. *Management of Biological Invasions*, 8, 153–161.
Firth L.B., Airoldi L., Bulleri F., Challinor S., Chee S.-Y., Evans A.J., Hanley M.E., Knights A.M., O'Shaughnessy K., Thompson R.C. & Hawkins S.J. (2020) Greening of grey infrastructure should not be used as a Trojan horse to facilitate coastal development. *Journal of Applied Ecology*, 57, 1762–1768.
Firth L.B., Thompson R.C., Bohn K., Abbiati M., Airoldi L., Bouma T.J., Bozzeda F., Ceccherelli V.U., Colangelo M.A., Evans A., Ferrario F., Hanley M.E., Hinz H., Hoggart S.P.G., Jackson J.E., Moore P., Morgan E.H., Perkol-Finkel S., Skov M.W., Strain E.M., van Belzen J. & Hawkins S.J. (2014) Between a rock and a hard place: environmental and engineering considerations when designing coastal defence structures. *Coastal Engineering*, 87, 122–135.
Hylkema A., Debrot A.O., Osinga R., Bron P.S., Heesink D.B., Izioka A.K., Reid C.B., Rippen J.C., Treibitz T., Yuval M. & Murk A.J. (2020) Fish assemblages of three common artificial reef designs during early colonization. *Ecological Engineering*, 157, 105994.
Mercader M., Blazy C., Di Pane J., Devissi C., Mercière A., Cheminée A., Thiriet P., Pastor J., Crec'hriou R., Verdoit-Jarraya M. & Lenfant P. (2019) Is artificial habitat diversity a key to restoring nurseries for juvenile coastal fish? *Ex situ* experiments on habitat selection and survival of juvenile seabreams. *Restoration Ecology*, 27, 1155–1165.
Nelson B.V. & Vance R.R. (1979) Diel foraging patterns of the sea urchin *Centrostephanus coronatus* as a predator avoidance strategy. *Marine Biology*, 51, 251–258.
Noordin Raja Omar R.M., Eng Kean C., Wagiman S., Mutalib Mat Hassan A., Hussein M., Bidin Raja Hassan R. & Omar Mat Hussin C. (1994) Design and construction of artificial reefs in Malaysia. *Bulletin of Marine Science*, 55, 1050–1061.
Sherrard T.R.W., Hawkins S.J., Barfield P., Kitou M., Bray S. & Osborne P.E. (2016) Hidden biodiversity in cryptic habitats provided by porous coastal defence structures. *Coastal Engineering*, 118, 12–20.

A study in 2009–2010 on a subtidal pipeline in a lagoon in the Mozambique Channel, Mayotte (1) reported that small swimthrough habitats created on pipeline anchor-weights, along with large swimthroughs and environmentally-sensitive material, were used by juvenile spiny lobster *Panulirus versicolor*, juvenile blue-and-yellow grouper *Epinephelus flavocaeruleus*, sea firs (Hydrozoa) and adult fishes from five families. After one month, juvenile spiny lobsters and blue-and-yellow groupers, sea firs, and adult damselfish/clownfish (Pomacentridae), wrasse (Labridae), butterflyfish (Chaetodontidae), squirrelfish/soldierfish (Holocentridae) and surgeonfish (Acanthuridae) were recorded on and around anchor-weights with swimthrough habitats and environmentally-sensitive material. Small swimthrough habitats were created by attaching basalt rocks or semi-cylindrical tiles to the horizontal surfaces of concrete anchor-weights placed over a seabed pipeline (400 mm diameter). Basalt may be considered an environmentally-sensitive material compared with concrete. Large swimthrough habitats were also created between the anchor-weights and pipeline. Habitat dimensions/numbers were not reported. A total of 260 anchor-weights were placed with one every 10 m along the pipeline at 0–26 m depth during December 2009–March 2010. Fishes were counted on and around the pipeline from videos after 1 month.

A replicated, paired sites, controlled study in 2013–2014 on subtidal seawalls and pontoons in five marinas in the Mediterranean Sea, France (2) found that creating small swimthrough habitats on seawalls and pontoons had mixed effects on juvenile seabream *Diplodus* spp. abundance and habitat usage on and around the structures, depending on the species, juvenile development stage, site and survey month. Over 17 months, juvenile seabream (four species) used swimthrough habitats created on seawalls as frequently as those created under pontoons, and in three of six comparisons, they used both more than seawall and pontoon surfaces without swimthroughs, but in the other three comparisons no significant difference was found (data reported as habitat preference index). Abundances on and around swimthroughs and seawall and pontoon surfaces varied depending on the species, development stage, site and survey month (swimthroughs: 0–6 individuals/survey for any one species; seawall and pontoon: both 0–2/survey; see paper for results). Small swimthrough habitats were created by attaching steel cages containing oyster shells (Biohuts: height: 0.8 m; length: 0.5 m; width: 0.3 m; mesh size: 25–50 mm) to seawalls and pontoons in March 2013. Eight Biohuts were attached to each of three vertical seawalls, and three were suspended under each of three pontoons, in each of five marinas (depth not reported). Biohuts were compared with seawall (height: 0.8 m; length: 5 m) and pontoon (4 m^2) surfaces without swimthroughs. Juvenile seabreams were counted on and around Biohuts and seawall/pontoon surfaces over 17 months.

A replicated, paired sites, controlled study in 2014 on three subtidal seawalls in a port in the Mediterranean Sea, France (3) found that creating small swimthrough habitats on seawalls had mixed effects on juvenile fish species richness, abundance and community composition on and around the walls, depending on the site, survey month and species. Over four months, at two of three sites, juvenile fish species richness and total abundance was higher on and around seawall surfaces with swimthrough habitats (3–4 species and 13–18 individuals/10 m seawall) than those without (0–1 species and 3–12 individuals/10 m). At the third site, there were no significant differences (1 species and 3 individuals/10 m seawall with and without swimthroughs). Community composition (data reported as statistical model results) and individual species abundances varied on and around seawall surfaces with and without swimthroughs, depending on the site, survey month and species (see paper for results). Six species recorded on and around swimthroughs were absent from seawall surfaces without. Small swimthrough habitats were created in May 2014 by attaching steel cages containing oyster shells (Biohuts) to seawall surfaces (30 m long). Thirty-five Biohuts (height: 0.8 m; length: 0.5 m; width: 0.3 m; mesh size: 25–50 mm) were attached at 1 m depth on each of three vertical seawalls. Biohuts were compared with adjacent seawall surfaces (30 m long) on each wall. Juvenile fishes were counted on and around seawall surfaces with and without Biohuts over four months.

A replicated study in 2014–2015 on subtidal pontoons in a marina in the Alboran Sea, Morocco (4) found that small swimthrough habitats created under pontoons were used by seven species of juvenile fishes. After 12 months, 34 juvenile mottled groupers (*Mycteroperca rubra*) and 28 juvenile dusky groupers (*Epinephelus marginatus*) were recorded on and around swimthrough habitats (Biohuts). Juveniles of three seabream species (*Diplodus sargus*, *Diplodus cervinus*, *Sarpa salpa*), European bass (*Dicentrarchus labrax*) and mullet (Mugilidae) were also recorded on and around swimthroughs. On average, there were 3 juveniles/Biohut. Small swimthrough habitats were created in June

2014 by attaching steel cages containing oyster shells (Biohuts) beneath pontoons. Fifty Biohuts (height: 0.8 m; length: 0.5 m; width: 0.3 m; mesh size: 25–50 mm) were attached at 1 m depth beneath pontoons (arrangement not reported). Juvenile fishes were counted on and around Biohuts after 12 months.

(1) Pioch S., Saussola P., Kilfoyle K. & Spieler R. (2011) Ecological design of marine construction for socio-economic benefits: ecosystem integration of a pipeline in a coral reef area. *Procedia Environmental Sciences*, 9, 148–152.
(2) Bouchoucha M., Darnaude A.M., Gudefin A., Neveu R., Verdoit-Jarraya M., Boissery P. & Lenfant P. (2016) Potential use of marinas as nursery grounds by rocky fishes: insights from four *Diplodus* species in the Mediterranean. *Marine Ecology Progress Series*, 547, 193–209.
(3) Mercader M., Mercière A., Saragoni G., Cheminée A., Crec'hriou R., Pastor J., Rider M., Dubas R., Lecaillon G., Boissery P. & Lenfant P. (2017) Small artificial habitats to enhance the nursery function for juvenile fish in a large commercial port of the Mediterranean. *Ecological Engineering*, 105, 78–86.
(4) Selfati M., El Ouamari N., Lenfant P., Fontcuberta A., Lecaillon G., Mesfioui A., Boissery P. & Bazairi H. (2018) Promoting restoration of fish communities using artificial habitats in coastal marinas. *Biological Conservation*, 219, 89–95.

3.9. Create large adjoining cavities or 'swimthrough' habitats (>100 mm) on subtidal artificial structures

- **Two studies** examined the effects of creating large adjoining cavities or 'swimthrough' habitats on subtidal artificial structures on the biodiversity of those structures. One study was in a lagoon in Mayotte[1] and one was in a marina in southeast USA[2].

COMMUNITY RESPONSE (1 STUDY)

- **Fish community composition (1 study):** One replicated, paired sites, controlled study in the USA[2] reported that large swimthrough habitats created in front of a subtidal artificial structure supported fish species that were absent from structure surfaces without swimthroughs.
- **Fish richness/diversity (1 study):** One replicated, paired sites, controlled study in the USA[2] found that creating large swimthrough habitats in front of a subtidal artificial structure increased the overall fish species richness on and around structure surfaces, but that effects varied depending on the fish size class.

POPULATION RESPONSE (1 STUDY)

- **Fish abundance (1 study):** One replicated, paired sites, controlled study in the USA[2] found that creating large swimthrough habitats in front of a subtidal artificial structure increased the overall fish abundance on and around structure surfaces, but that individual species abundances varied depending on the species, size class and survey month.

BEHAVIOUR (1 STUDY)

- **Use (1 study):** One study in Mayotte[1] reported that large swimthrough habitats created on a subtidal artificial structure, along with small swimthroughs and environmentally-sensitive material, were used by juvenile spiny lobsters and groupers, sea firs, and adult fishes from five families.

Background

Large adjoining cavities or 'swimthrough' habitats are not well-studied in subtidal rocky habitats. They may form through weathering of softer rocks, amongst loosely-consolidated boulders, or within three-dimensional structures created by organisms. They likely provide organisms refuge from predation, in the same way crevice and hole habitats do (Mercader *et al.* 2019; Nelson & Vance 1979). They could also serve as corridors, connecting adjacent refuge habitats. The size and density of cavities or swimthroughs is likely to affect the size, abundance and variety of organisms that can use them. Small habitats can provide refuge for small-bodied organisms but may exclude larger organisms, limit their growth and get rapidly filled-up (Firth *et al.* 2020). Large habitats can be used by larger-bodied organisms but may not provide sufficient refuge from predators for smaller organisms. By default, cavities and swimthroughs contain shaded surfaces, which can be associated with the presence of non-native species (Dafforn 2017).

Cavities/swimthroughs are sometimes present on marine artificial structures made of consolidated boulders or blocks (Sherrard *et al.* 2016) or gabion baskets (Firth *et al.* 2014), but are absent from many other structures. Large adjoining cavities or 'swimthrough' habitats can be created on subtidal artificial structures by adding or removing material, either during construction or retrospectively.

There is a body of literature investigating the effects of creating swimthrough habitats on artificial reefs (Brotto *et al.* 2006; Hylkema *et al.* 2020; Sherman *et al.* 2002). These studies are not included in this synopsis, which focusses on *in situ* conservation actions to enhance the biodiversity of structures that are engineered to fulfil a primary function other than providing artificial habitats.

Definition: 'Large adjoining cavities or 'swimthrough' habitats' are adjoining internal cavities sheltered from, but with access to/from, outside the structure. Dimensions can vary but are >100 mm in any direction.

See also: *Create hole habitats (>50 mm) on subtidal artificial structures*; *Create crevice habitats (>50 mm) on subtidal artificial structures*; *Create small adjoining cavities or 'swimthrough' habitats (≤100 mm) on subtidal artificial structures*.

Brotto D.S., Krohling W., Brum S. & Zalmon I.R. (2006) Usage patterns of an artificial reef by the fish community on the northern coast of Rio de Janeiro – Brazil. *Journal of Coastal Research*, 39, 1276–1280.

Dafforn K.A. (2017) Eco-engineering and management strategies for marine infrastructures to reduce establishment and dispersal of non-indigenous species. *Management of Biological Invasions*, 8, 153–161.

Firth L.B., Airoldi L., Bulleri F., Challinor S., Chee S.-Y., Evans A.J., Hanley M.E., Knights A.M., O'Shaughnessy K., Thompson R.C. & Hawkins S.J. (2020) Greening of grey infrastructure should not be used as a Trojan horse to facilitate coastal development. *Journal of Applied Ecology*, 57, 1762–1768.

Firth L.B., Thompson R.C., Bohn K., Abbiati M., Airoldi L., Bouma T.J., Bozzeda F., Ceccherelli V.U., Colangelo M.A., Evans A., Ferrario F., Hanley M.E., Hinz H., Hoggart S.P.G., Jackson J.E., Moore P., Morgan E.H., Perkol-Finkel S., Skov M.W., Strain E.M., van Belzen J. & Hawkins S.J. (2014) Between a rock and a hard place: environmental and engineering considerations when designing coastal defence structures. *Coastal Engineering*, 87, 122–135.

Hylkema A., Debrot A.O., Osinga R., Bron P.S., Heesink D.B., Izioka A.K., Reid C.B., Rippen J.C., Treibitz T., Yuval M. & Murk A.J. (2020) Fish assemblages of three common artificial reef designs during early colonization. *Ecological Engineering*, 157, 105994.

Mercader M., Blazy C., Di Pane J., Devissi C., Mercière A., Cheminée A., Thiriet P., Pastor J., Crec'hriou R., Verdoit-Jarraya M. & Lenfant P. (2019) Is artificial habitat diversity a key to restoring nurseries for juvenile coastal fish? *Ex situ* experiments on habitat selection and survival of juvenile seabreams. *Restoration Ecology*, 27, 1155–1165.

Nelson B.V. & Vance R.R. (1979) Diel foraging patterns of the sea urchin *Centrostephanus coronatus* as a predator avoidance strategy. *Marine Biology*, 51, 251–258.

Sherrard T.R.W., Hawkins S.J., Barfield P., Kitou M., Bray S. & Osborne P.E. (2016) Hidden biodiversity in cryptic habitats provided by porous coastal defence structures. *Coastal Engineering*, 118, 12–20.

Sherman R.L., Gilliam D.S. & Spieler R.E. (2002) Artificial reef design: void space, complexity, and attractants. *ICES Journal of Marine Science*, 59, S196–S200.

A study in 2009–2010 on a subtidal pipeline in a lagoon in the Mozambique Channel, Mayotte (1) reported that large swimthrough habitats created on pipeline anchor-weights, along with small swimthroughs and environmentally-sensitive material, were used by juvenile spiny lobster *Panulirus versicolor*, juvenile blue-and-yellow grouper *Epinephelus flavocaeruleus*, sea firs (Hydrozoa), and adult fishes from five families. After one month, juvenile spiny lobsters and blue-and-yellow groupers, sea firs, and adult damselfish/clownfish (Pomacentridae), wrasse (Labridae), butterflyfish (Chaetodontidae), squirrelfish/soldierfish (Holocentridae) and surgeonfish (Acanthuridae) were recorded on and around anchor-weights with swimthroughs and environmentally-sensitive material. Large swimthrough habitats were created by leaving gaps between concrete anchor-weights placed over a seabed pipeline (400 mm diameter). Anchor-weights also had basalt rocks or semi-cylindrical tiles attached to the top, creating small swimthrough habitats. Basalt may be considered an environmentally-sensitive material compared with concrete. Habitat dimensions/numbers were not reported. A total of 260 anchor-weights were placed with one every 10 m along the pipeline at 0–26 m depth during December 2009–March 2010. Fishes were counted on and around the pipeline from videos after 1 month.

A replicated, paired sites, controlled study in 2015–2016 on a seawall in a marina in Port Everglades, USA (2) found that creating large swimthrough habitats in front of the seawall increased the fish species richness and abundance on and around seawall surfaces, but that effects varied depending on the species, size class and survey month. Over 14 months, total fish abundance was higher on and around seawall surfaces with swimthroughs (1,614 individuals) than those without (655 individuals). Fish species richness and average abundance (all size classes combined) was also higher (swimthroughs: 4 species and 10 individuals/survey; no swimthroughs: 2 species and 4 individuals/survey). This was also true for fishes in 20–300 mm size classes (swimthroughs: 0–2 species and 1–3 individuals/survey; no swimthroughs: 0–1 species and individuals/survey), but not for smaller or larger groups (both 0 species/survey; swimthroughs: 0–1 individuals/survey; no swimthroughs: 0 individuals/survey). Species abundances around seawall surfaces with and without swimthroughs varied depending on the species, size class and survey month (see paper for results). Sixteen species recorded on and around swimthroughs were absent from seawall surfaces without. Large swimthrough habitats (length: ~510 mm; width: ~250 mm; height: ~100 mm) were created by placing concrete bricks as spacers between four horizontally-stacked concrete paving slabs (510 × 510 mm). Twelve stacks of pavers with three swimthroughs/stack were placed at 1–3 m depth on silty seabed 0.5 m in front of a seawall in February 2015. Fishes were counted on and around sections of the seawall (1.5 × 1.5 m) with and without swimthroughs over 14 months.

(1) Pioch S., Saussola P., Kilfoyle K. & Spieler R. (2011) Ecological design of marine construction for socio-economic benefits: ecosystem integration of a pipeline in a coral reef area. *Procedia Environmental Sciences*, 9, 148–152.
(2) Patranella A., Kilfoyle K., Pioch S. & Spieler R.E. (2017) Artificial reefs as juvenile fish habitat in a marina. *Journal of Coastal Research*, 33, 1341–1351.

3.10. Create small protrusions (1–50 mm) on subtidal artificial structures

- **One study** examined the effects of creating small protrusions on subtidal artificial structures on the biodiversity of those structures. The study was on an open coastline in Japan[1].

COMMUNITY RESPONSE (0 STUDIES)

POPULATION RESPONSE (1 STUDY)

- **Algal abundance (1 study):** One controlled study in Japan[1] reported that creating small protrusions on a subtidal artificial structure had mixed effects on the macroalgal abundance on structure surfaces, depending on the species.

BEHAVIOUR (0 STUDIES)

Background

Small protrusions create vertical or horizontal (i.e. overhangs) relief in subtidal rocky habitats. They can provide organisms refuge from predation or grazing (Wahl & Hoppe 2002) and have positive effects on fish populations (Morris *et al.* 2018). Some species preferentially recruit to habitats with high vertical or horizontal relief (Andrews & Anderson 2004). The size and density of protrusions is likely to affect the size, abundance and variety of organisms that can use them. Small habitats can provide refuge for small-bodied organisms but may exclude larger organisms and limit their growth. Large habitats can be used by larger-bodied organisms but may not provide sufficient refuge from predators for smaller organisms. By default, horizontal protrusions (overhangs) create shaded and downward-facing surfaces, which can be associated with the presence of non-native species (Dafforn 2017).

Protrusions are sometimes present on artificial structures such as cable mattresses (Lacey & Hayes 2020) or quarried boulders (MacArthur *et al.* 2020), but are often absent from other types of structures. Small protrusions can be created on subtidal artificial structures by adding material, either during construction or retrospectively.

There is a body of literature investigating the effects of creating protrusions on artificial reefs (e.g. Gratwicke & Speight 2005; Morris *et al.* 2018). These studies are not included in this synopsis, which focusses on *in situ* conservation actions to enhance the biodiversity of structures that are engineered to fulfil a primary function other than providing artificial habitats.

Definition: 'Small protrusions' are elevations with a length to width ratio ≤3:1 that protrude 1–50 mm from the substratum (modified from "Small elevations" in Strain *et al.* 2018).

See also: *Create textured surfaces (≤1 mm) on subtidal artificial structures*; *Create natural rocky reef topography on subtidal artificial structures*; *Create large protrusions (>50 mm) on subtidal artificial structures*; *Create small ridges or ledges (1–50 mm) on subtidal artificial structures*; *Create large ridges or ledges (>50 mm) on subtidal artificial structures*; *Create groove habitats* and *small protrusions, ridges or ledges (1–50 mm) on subtidal artificial structures*.

Andrews K.S. & Anderson T.W. (2004) Habitat-dependent recruitment of two temperate reef fishes at multiple spatial scales. *Marine Ecology Progress Series*, 277, 231–244.

Dafforn K.A. (2017) Eco-engineering and management strategies for marine infrastructures to reduce establishment and dispersal of non-indigenous species. *Management of Biological Invasions*, 8, 153–161.

Gratwicke B. & Speight M.R. (2005) Effects of habitat complexity on Caribbean marine fish assemblages. *Marine Ecology Progress Series*, 292, 301–310.

Lacey N.C. & Hayes P. (2020) Epifauna associated with subsea pipelines in the North Sea. *ICES Journal of Marine Science*, 77, 1137–1147.

MacArthur M., Naylor L.A., Hansom J.D. & Burrows M.T. (2020) Ecological enhancement of coastal engineering structures: passive enhancement techniques. *Science of the Total Environment*, 740, 139981.

Morris R.L., Porter A.G., Figueira W.F., Coleman R.A., Fobert E.K. & Ferrari R. (2018) Fish-smart seawalls: a decision tool for adaptive management of marine infrastructure. *Frontiers in Ecology and the Environment*, 16, 278–287.

Strain E.M.A., Olabarria C., Mayer-Pinto M., Cumbo V., Morris R.L., Bugnot A.B., Dafforn K.A., Heery E., Firth L.B., Brooks P.R. & Bishop M.J. (2018) Eco-engineering urban infrastructure for marine and coastal biodiversity: which interventions have the greatest ecological benefit? *Journal of Applied Ecology*, 55, 426–441.

Wahl M. & Hoppe K. (2002) Interactions between substratum rugosity, colonization density and periwinkle grazing efficiency. *Marine Ecology Progress Series*, 225, 239–249.

A controlled study in 1985–1989 on a subtidal breakwater block on open coastline in Toyama Bay, Japan (1) reported that small protrusions created on the block supported more kelp *Ecklonia stolonifera* but less canopy algae *Sargassum* spp. than a block surface without protrusions. Data were not statistically tested. After 42 months, there were 58 kelp individuals on the surface with small protrusions (wet weight: 1.09 kg) and 20 on the surface without (0.31 kg). There were 2–3 individuals of each of three other canopy algae species on the surface with protrusions (0.01–0.09 kg) and 3–18 of each on the surface without (0.05–0.17 kg). Small protrusions were created on a concrete breakwater block (2.3 × 2.3 × 0.8 m) by attaching 45 pebbles (diameter/height: 35–45 mm), evenly-spaced on a 644 × 529 mm horizontal surface. One adjacent surface had no protrusions. The block was placed on sandy seabed at 9 m depth in November 1985. Macroalgae on surfaces with and without small protrusions were counted and weighed (wet weight) after 42 months.

(1) Watanuki A. & Yamamoto H. (1990) Settlement of seaweeds on coastal structures. *Hydrobiologia*, 204, 275–280.

3.11. Create large protrusions (>50 mm) on subtidal artificial structures

- We found no studies that evaluated the effects of creating large protrusions on subtidal artificial structures on the biodiversity of those structures.

This means we did not find any studies that directly evaluated this intervention during our literature searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Background

Large protrusions create vertical or horizontal (i.e. overhangs) relief in subtidal rocky habitats. They can provide organisms refuge from predation (Meese & Lowe 2020) and have positive effects on fish populations (Morris *et al.* 2018). Some species preferentially recruit to habitats with high vertical or horizontal relief (Andrews & Anderson 2004). The size and density of protrusions is likely to affect the size, abundance and variety of organisms that can use them. Small habitats can provide refuge for small-bodied organisms but may exclude larger organisms and limit their growth. Large habitats can be used by larger-bodied organisms but may not provide sufficient refuge from predators for smaller organisms. By default, horizontal protrusions (overhangs) create shaded and downward-facing surfaces, which can be associated with the presence of non-native species (Dafforn 2017).

Protrusions are sometimes present on quarried boulders used in marine artificial structures (MacArthur *et al.* 2020), but are often absent from other types of structures. Large protrusions can be created on subtidal artificial structures by adding material, either during construction or retrospectively.

There is a body of literature investigating the effects of creating protrusions on artificial reefs (e.g. Gratwicke & Speight 2005; Morris *et al.* 2018). These studies are not included in this synopsis, which focusses on *in situ* conservation actions to enhance the biodiversity of structures that are engineered to fulfil a primary function other than providing artificial habitats.

Definition: 'Large protrusions' are elevations with a length to width ratio ≤3:1 that protrude >50 mm from the substratum (modified from "Large elevations" in Strain *et al.* 2018).

See also: *Create textured surfaces (≤1 mm) on subtidal artificial structures*; *Create natural rocky reef topography on subtidal artificial structures*; *Create small protrusions (1–50 mm) on subtidal artificial structures*; *Create small ridges or ledges (1–50 mm) on subtidal artificial structures*; *Create large ridges or ledges (>50 mm) on subtidal artificial structures*; *Create groove habitats* and *small protrusions, ridges or ledges (1–50 mm) on subtidal artificial structures*.

Andrews K.S. & Anderson T.W. (2004) Habitat-dependent recruitment of two temperate reef fishes at multiple spatial scales. *Marine Ecology Progress Series*, 277, 231–244.

Dafforn K.A. (2017) Eco-engineering and management strategies for marine infrastructures to reduce establishment and dispersal of non-indigenous species. *Management of Biological Invasions*, 8, 153–161.

Gratwicke B. & Speight M.R. (2005) Effects of habitat complexity on Caribbean marine fish assemblages. *Marine Ecology Progress Series*, 292, 301–310.

MacArthur M., Naylor L.A., Hansom J.D. & Burrows M.T. (2020) Ecological enhancement of coastal engineering structures: passive enhancement techniques. *Science of the Total Environment*, 740, 139981.

Meese E.N. & Lowe C.G. (2020) Environmental effects on daytime sheltering behaviours of California horn sharks (*Heterodontus francisci*). *Environmental Biology of Fishes*, 103, 703–717.

Morris R.L., Porter A.G., Figueira W.F., Coleman R.A., Fobert E.K. & Ferrari R. (2018) Fish-smart seawalls: a decision tool for adaptive management of marine infrastructure. *Frontiers in Ecology and the Environment*, 16, 278–287.

Strain E.M.A., Olabarria C., Mayer-Pinto M., Cumbo V., Morris R.L., Bugnot A.B., Dafforn K.A., Heery E., Firth L.B., Brooks P.R. & Bishop M.J. (2018) Eco-engineering urban infrastructure for marine and coastal biodiversity: which interventions have the greatest ecological benefit? *Journal of Applied Ecology*, 55, 426–441.

3.12. Create small ridges or ledges (1–50 mm) on subtidal artificial structures

- We found no studies that evaluated the effects of creating small ridges or ledges on subtidal artificial structures on the biodiversity of those structures.

This means we did not find any studies that directly evaluated this intervention during our literature searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Background

Small ridges and ledges create vertical or horizontal (i.e. overhangs) relief in subtidal rocky habitats. They can provide organisms refuge from predation or grazing (Wahl & Hoppe 2002) and have positive effects on fish populations (Morris *et al.* 2018). Some species preferentially recruit to habitats with high vertical or horizontal relief (Andrews & Anderson 2004). The size and density of ridges and ledges is likely to affect the size, abundance and variety of organisms that can use them. Small habitats can provide refuge for small-bodied organisms but may exclude larger organisms and limit their growth.

Large habitats can be used by larger-bodied organisms but may not provide sufficient refuge from predators for smaller organisms. By default, horizontal ledges (overhangs) create shaded and downward-facing surfaces, which can be associated with the presence of non-native species (Dafforn 2017).

Ridges and ledges are sometimes present on artificial structures such as cable mattresses (Lacey & Hayes 2020) or quarried boulders (MacArthur *et al.* 2020), but are often absent from other types of structures. Small ridges and ledges can be created on subtidal artificial structures by adding material, either during construction or retrospectively.

There is a body of literature investigating the effects of creating these habitats on artificial reefs (e.g. Gratwicke & Speight 2005; Morris *et al.* 2018). These studies are not included in this synopsis, which focusses on *in situ* conservation actions to enhance the biodiversity of structures that are engineered to fulfil a primary function other than providing artificial habitats.

Definition: 'Small ridges and ledges' are elevations with a length to width ratio >3:1 that protrude 1–50 mm from the substratum (modified from "Small elevations" in Strain *et al.* 2018). On vertical surfaces, vertically-orientated elevations that fit these criteria are referred to as 'ridges', while horizontal ones are referred to as 'ledges'. On horizontal surfaces, these features are referred to as 'ridges' regardless of their orientation.

See also: *Create textured surfaces (≤1 mm) on subtidal artificial structures*; *Create natural rocky reef topography on subtidal artificial structures*; *Create small protrusions (1–50 mm) on subtidal artificial structures*; *Create large protrusions (>50 mm) on subtidal artificial structures*; *Create large ridges or ledges (>50 mm) on subtidal artificial structures*; *Create groove habitats* and *small protrusions, ridges or ledges (1–50 mm) on subtidal artificial structures*.

Andrews K.S. & Anderson T.W. (2004) Habitat-dependent recruitment of two temperate reef fishes at multiple spatial scales. *Marine Ecology Progress Series*, 277, 231–244.

Dafforn K.A. (2017) Eco-engineering and management strategies for marine infrastructures to reduce establishment and dispersal of non-indigenous species. *Management of Biological Invasions*, 8, 153–161.

Gratwicke B. & Speight M.R. (2005) Effects of habitat complexity on Caribbean marine fish assemblages. *Marine Ecology Progress Series*, 292, 301–310.

Lacey N.C. & Hayes P. (2020) Epifauna associated with subsea pipelines in the North Sea. *ICES Journal of Marine Science*, 77, 1137–1147.

MacArthur M., Naylor L.A., Hansom J.D. & Burrows M.T. (2020) Ecological enhancement of coastal engineering structures: passive enhancement techniques. *Science of the Total Environment*, 740, 139981.

Morris R.L., Porter A.G., Figueira W.F., Coleman R.A., Fobert E.K. & Ferrari R. (2018) Fish-smart seawalls: a decision tool for adaptive management of marine infrastructure. *Frontiers in Ecology and the Environment*, 16, 278–287.

Strain E.M.A., Olabarria C., Mayer-Pinto M., Cumbo V., Morris R.L., Bugnot A.B., Dafforn K.A., Heery E., Firth L.B., Brooks P.R. & Bishop M.J. (2018) Eco-engineering urban infrastructure for marine and coastal biodiversity: which interventions have the greatest ecological benefit? *Journal of Applied Ecology*, 55, 426–441.

Wahl M. & Hoppe K. (2002) Interactions between substratum rugosity, colonization density and periwinkle grazing efficiency. *Marine Ecology Progress Series*, 225, 239–249.

3.13. Create large ridges or ledges (>50 mm) on subtidal artificial structures

- We found no studies that evaluated the effects of creating large ridges or ledges on subtidal artificial structures on the biodiversity of those structures.

This means we did not find any studies that directly evaluated this intervention during our literature searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Background

Large ridges and ledges create vertical or horizontal (i.e. overhangs) relief in subtidal rocky habitats. They can provide organisms refuge from predation (Meese & Lowe 2020) and have positive effects on fish populations (Morris *et al.* 2018). Some species preferentially recruit to habitats with high vertical or horizontal relief (Andrews & Anderson 2004). The size and density of ridges and ledges is likely to affect the size, abundance and variety of organisms that can use them. Small habitats can provide refuge for small-bodied organisms but may exclude larger organisms and limit their growth. Large habitats can be used by larger-bodied organisms but may not provide sufficient refuge from predators for smaller organisms. By default, horizontal ledges (overhangs) create shaded and downward-facing surfaces, which can be associated with the presence of non-native species (Dafforn 2017).

Ridges and ledges are sometimes present on quarried boulders used in marine artificial structures (MacArthur *et al.* 2020), but are often absent from other types of structures. Large ridges and ledges can be created on subtidal artificial structures by adding material, either during construction or retrospectively.

There is a body of literature investigating the effects of creating these habitats on artificial reefs (e.g. Gratwicke & Speight 2005; Morris *et al.* 2018). These studies are not included in this synopsis, which focusses on *in situ* conservation actions to enhance the biodiversity of structures that are engineered to fulfil a primary function other than providing artificial habitats.

Definition: 'Large ridges and ledges' are elevations with a length to width ratio >3:1 that protrude >50 mm from the substratum (modified from "Large elevations" in Strain *et al.* 2018). On vertical surfaces, vertically-orientated elevations that fit these criteria are referred to as 'ridges', while horizontal ones are referred to as 'ledges'. On horizontal surfaces, these features are referred to as 'ridges' regardless of their orientation.

See also: *Create textured surfaces (≤1 mm) on subtidal artificial structures*; *Create natural rocky reef topography on subtidal artificial structures*; *Create small protrusions (1–50 mm)* *on subtidal artificial structures*; *Create large protrusions (>50 mm) on subtidal artificial structures*; *Create small ridges or ledges (1–50 mm) on subtidal artificial structures*; *Create groove habitats* and *small protrusions, ridges or ledges (1–50 mm) on subtidal artificial structures*.

Andrews K.S. & Anderson T.W. (2004) Habitat-dependent recruitment of two temperate reef fishes at multiple spatial scales. *Marine Ecology Progress Series*, 277, 231–244.

Dafforn K.A. (2017) Eco-engineering and management strategies for marine infrastructures to reduce establishment and dispersal of non-indigenous species. *Management of Biological Invasions*, 8, 153–161.

Gratwicke B. & Speight M.R. (2005) Effects of habitat complexity on Caribbean marine fish assemblages. *Marine Ecology Progress Series*, 292, 301–310.

MacArthur M., Naylor L.A., Hansom J.D. & Burrows M.T. (2020) Ecological enhancement of coastal engineering structures: passive enhancement techniques. *Science of the Total Environment*, 740, 139981.

Meese E.N. & Lowe C.G. (2020) Environmental effects on daytime sheltering behaviours of California horn sharks (*Heterodontus francisci*). *Environmental Biology of Fishes*, 103, 703–717.

Morris R.L., Porter A.G., Figueira W.F., Coleman R.A., Fobert E.K. & Ferrari R. (2018) Fish-smart seawalls: a decision tool for adaptive management of marine infrastructure. *Frontiers in Ecology and the Environment*, 16, 278–287.

Strain E.M.A., Olabarria C., Mayer-Pinto M., Cumbo V., Morris R.L., Bugnot A.B., Dafforn K.A., Heery E., Firth L.B., Brooks P.R. & Bishop M.J. (2018) Eco-engineering urban infrastructure for marine and coastal biodiversity: which interventions have the greatest ecological benefit? *Journal of Applied Ecology*, 55, 426–441.

3.14. Create grooves *and* small protrusions, ridges or ledges (1–50 mm) on subtidal artificial structures

- **Three studies** examined the effects of creating groove habitats *and* small protrusions, ridges or ledges on subtidal artificial structures on the biodiversity of those structures. Two studies were in marinas in northern Israel[1] and the UK[2] and one was on an open coastline in southeast Spain[3].

COMMUNITY RESPONSE (2 STUDIES)

- **Overall community composition (2 studies):** One of two replicated, randomized, controlled studies (including one paired sites, before-and-after study) in Israel[1] and the UK[2] found that groove habitats and small ledges created on a subtidal artificial structure, along with holes and environmentally-sensitive material, altered the combined macroalgae and invertebrate community composition on structure surfaces[1]. They also supported non-mobile invertebrate species that were absent from structure surfaces without added habitat features. One study[2] found that creating grooves and small protrusions had mixed effects on the community composition, depending on the orientation of structure surfaces.
- **Overall richness/diversity (2 studies):** One of two replicated, randomized, controlled studies (including one paired sites, before-and-after study) in Israel[1] and the UK[2] found that creating groove habitats and small ledges on a subtidal artificial structure, along with holes and environmentally-sensitive material, increased the combined macroalgae and invertebrate species richness and diversity on structure surfaces[1]. One study[2] found that creating grooves and small protrusions did not increase the species diversity but had mixed effects on species richness, depending on the orientation of structure surfaces.

POPULATION RESPONSE (1 STUDY)

- **Overall abundance (1 study):** One replicated, randomized, controlled study in the UK[2] found that creating groove habitats and small protrusions on subtidal artificial structures had mixed effects on the combined macroalgae and non-mobile invertebrate abundance, depending on the orientation of structure surfaces.

BEHAVIOUR (1 STUDY)

- **Use (1 study):** One replicated study in Spain[3] reported that groove habitats and small protrusions created on subtidal artificial structures were colonized by macroalgae and non-mobile invertebrates.

Background

Grooves, small protrusions, ridges and ledges provide organisms refuge from predation or grazing in subtidal rocky habitats (Nelson & Vance 1979; Wahl & Hoppe 2002). Some species preferentially settle in and around them (Andrews & Anderson 2004; Bourget *et al.* 1994). The size and density of grooves, protrusions, ridges and ledges is likely to affect the size, abundance and variety of organisms that can use them. Small habitats can provide refuge for small-bodied organisms but may exclude larger organisms, limit their growth and get rapidly filled-up (Firth *et al.* 2020). Large habitats can be used by larger-bodied organisms but may not provide sufficient refuge from predators for smaller organisms. By default, horizontal protrusions/ledges (overhangs) create shaded and downward-facing surfaces, which can be associated with the presence of non-native species (Dafforn 2017).

Grooves, protrusions, ridges and ledges are sometimes present on artificial structures such as cable mattresses (Lacey & Hayes 2020) or quarried boulders (MacArthur *et al.* 2020), but are often absent from other types of structures. Groove habitats, small protrusions, ridges and ledges can be created on subtidal artificial structures by adding or removing material, either during construction or retrospectively. In some scenarios, creating one will automatically result in creation of the other (i.e. grooves created in between created protrusions/ridges/ledges, or *vice versa*). Studies containing such scenarios are considered under this joint intervention.

Definition: 'Groove habitats' are depressions with a length to width ratio >3:1 and depth 1–50 mm (modified from "Crevices" in Strain *et al.* 2018). 'Small protrusions' are elevations with a length to width ratio ≤3:1 that protrude 1–50 mm from the substratum (modified from "Small elevations" in Strain *et al.* 2018). 'Small ridges and ledges' are elevations with a length to width ratio >3:1 that protrude 1–50 mm from the substratum (modified from "Small elevations" in Strain *et al.* 2018). On vertical surfaces, vertically-orientated elevations that fit these criteria are referred to as 'ridges', while horizontal ones are referred to as 'ledges'. On horizontal surfaces, these features are referred to as 'ridges' regardless of their orientation.

See also: *Create textured surfaces (≤1 mm) on subtidal artificial structures*; *Create natural rocky reef topography on subtidal artificial structures*; *Create pit habitats (1–50 mm) on subtidal artificial structures*; *Create hole habitats (>50 mm) on subtidal artificial structures*; *Create groove habitats (1–50 mm) on subtidal artificial structures*; *Create crevice habitats (>50 mm) on subtidal artificial structures*; *Create small protrusions (1–50 mm) on subtidal artificial structures*; *Create small ridges or ledges (1–50 mm) on subtidal artificial structures*.

Andrews K.S. & Anderson T.W. (2004) Habitat-dependent recruitment of two temperate reef fishes at multiple spatial scales. *Marine Ecology Progress Series*, 277, 231–244.

Bourget E., DeGuise J. & Daigle G. (1994) Scales of substratum heterogeneity, structural complexity, and the early establishment of a marine epibenthic community. *Journal of Experimental Marine Biology and Ecology*, 181, 31–51.

Dafforn K.A. (2017) Eco-engineering and management strategies for marine infrastructures to reduce establishment and dispersal of non-indigenous species. *Management of Biological Invasions*, 8, 153–161.

Firth L.B., Airoldi L., Bulleri F., Challinor S., Chee S.-Y., Evans A.J., Hanley M.E., Knights A.M., O'Shaughnessy K., Thompson R.C. & Hawkins S.J. (2020) Greening of grey infrastructure should not be used as a Trojan horse to facilitate coastal development. *Journal of Applied Ecology*, 57, 1762–1768.

Lacey N.C. & Hayes P. (2020) Epifauna associated with subsea pipelines in the North Sea. *ICES Journal of Marine Science*, 77, 1137–1147.

MacArthur M., Naylor L.A., Hansom J.D. & Burrows M.T. (2020) Ecological enhancement of coastal engineering structures: passive enhancement techniques. *Science of the Total Environment*, 740, 139981.

Nelson B.V. & Vance R.R. (1979) Diel foraging patterns of the sea urchin *Centrostephanus coronatus* as a predator avoidance strategy. *Marine Biology*, 51, 251–258.

Strain E.M.A., Olabarria C., Mayer-Pinto M., Cumbo V., Morris R.L., Bugnot A.B., Dafforn K.A., Heery E., Firth L.B., Brooks P.R. & Bishop M.J. (2018) Eco-engineering urban infrastructure for marine and coastal biodiversity: which interventions have the greatest ecological benefit? *Journal of Applied Ecology*, 55, 426–441.

Wahl M. & Hoppe K. (2002) Interactions between substratum rugosity, colonization density and periwinkle grazing efficiency. *Marine Ecology Progress Series*, 225, 239–249.

A replicated, randomized, paired sites, controlled, before-and-after study in 2014–2016 on a subtidal seawall in a marina in the Mediterranean Sea, Israel (1) found that groove habitats and small ledges created on seawall panels, along with holes and environmentally-sensitive material, supported higher macroalgae and invertebrate species diversity and richness and different community composition compared with standard-concrete seawall surfaces without added habitats. After 22 months, macroalgae and invertebrate species diversity (data reported as Shannon index) and richness was higher on panels with added habitats (9 species/quadrat) than on seawall surfaces without (5/quadrat), and compared with seawall surfaces before habitats were added (1/quadrat). Community composition differed between panels with added habitats and seawall surfaces without (data reported as statistical model results). Two non-mobile invertebrate species groups recorded on panels were absent from surfaces without. It is not clear whether these effects were the direct result of creating grooves and ledges, holes, or using environmentally-sensitive material. Groove habitats and small ledges were created on seawall panels (height: 1.5 m; width: 0.9 m; thickness: 130 mm) using a formliner. Each panel had multiple interlocking rectangular grooves and ledges (length: 50–150 mm; width/depth/height: 10–50 mm) amongst multiple holes. Panels were made from patented ECOncrete™ material. Four panels were attached to a vertical concrete seawall in November 2014. The bottom 1.2 m were subtidal. Seawall surfaces were subtidal areas of seawall cleared of organisms (height: 1.2 m; width: 0.9 m) adjacent to each panel. Macroalgae and invertebrates were counted in one 300 × 300 mm randomly-placed quadrat on each panel and seawall surface over 22 months.

A replicated, randomized, controlled study in 2016 on pontoons in a marina in the Fal estuary, UK (2) found that upward-facing settlement plates with groove habitats and small protrusions supported different macroalgae and invertebrate community composition, with similar species diversity but higher species richness and abundances, than upward-facing plates without grooves or protrusions, but that there were no significant differences on downward-facing plates. After six months, upward-facing plates with grooves and protrusions supported different macroalgae and invertebrate community composition (data reported as statistical model results) with similar species diversity (data not reported) but higher species richness (20 species/plate, reported from Figure 4) and macroalgae and non-mobile invertebrate live cover (29% cover), compared with plates without grooves and protrusions (15 species/plate, 13% cover). On downward-facing plates, there were no significant differences between plates with and without grooves and protrusions (both 25 species/plate; 92 vs 86% cover). Settlement plates (150 × 150 mm) were moulded with a regular grid of six groove habitats (length: 150 mm; width/depth: 10 mm) between 15 rectangular small protrusions (length: ~44 mm; width: ~22 mm; height: 10 mm) on one surface, but flat on the other. Plates were either standard-concrete or oyster-shell-concrete. Forty plates were suspended horizontally, randomly arranged, beneath floating pontoons at 2–3 m depth in April 2016. Ten of each material had grooves and protrusions facing up, while 10 of each faced down. Macroalgae and invertebrates on upward- and downward-facing surfaces were counted in the laboratory over six months.

A replicated study in 2014–2015 on a subtidal rocky reef on open coastline in the Alboran Sea, Spain (3) reported that settlement plates with groove habitats and small protrusions supported 33 macroalgae and non-mobile invertebrate species groups. After 11 months, plates with grooves and protrusions supported 33 species groups in total (20 macroalgae, 13 non-mobile invertebrates). On average, there were nine species/pair of plates, with 55% live cover. Settlement plates (170 × 170 mm) were cut to create a regular grid of six groove habitats (length: 170 mm; width/depth: ~7 mm) between 16 square protrusions (length/width: 30 mm; height: ~7 mm) on their surfaces. Plates were either sandstone, limestone, gabbro, slate or concrete. Two of each material were attached horizontally at 15 m depth on gneiss rocky seabed in each of three sites in June 2014. Macroalgae and non-mobile invertebrates on each pair of plates were counted from photographs over 11 months.

(1) Perkol-Finkel S., Hadary T., Rella A., Shirazi R. & Sella I. (2018) Seascape architecture – incorporating ecological considerations in design of coastal and marine infrastructure. *Ecological Engineering*, 120, 645–654.

(2) Hanlon N., Firth L.B. & Knights A.M. (2018) Time-dependent effects of orientation, heterogeneity and composition determines benthic biological community recruitment patterns on subtidal artificial structures. *Ecological Engineering*, 122, 219–228.
(3) Sempere-Valverde J., Ostalé-Valriberas E., Farfán G.M. & Espinosa F. (2018) Substratum type affects recruitment and development of marine assemblages over artificial substrata: a case study in the Alboran Sea. *Estuarine, Coastal and Shelf Science*, 204, 56–65.

3.15. Create short flexible habitats (1–50 mm) on subtidal artificial structures

- **Three studies** examined the effects of creating short flexible habitats on subtidal artificial structures on the biodiversity of those structures. Two studies were in an estuary in southeast Australia[1,2] and one was in marinas in northwest France[3].

COMMUNITY RESPONSE (2 STUDIES)

- **Invertebrate community composition (2 studies):** Two replicated, randomized, controlled studies (including one paired sites study) in Australia[2] and France[3] found that creating short flexible habitats on subtidal artificial structures had mixed effects on the mobile[2,3] and/or non-mobile[3] invertebrate community composition, depending on the density[3] or length[2] of flexible habitats and/or the site[3]. One of the studies[2] found it altered the non-mobile invertebrate community composition.
- **Invertebrate richness/diversity (1 study):** One replicated, randomized, paired sites, controlled study in France[3] found that creating short flexible habitats on subtidal artificial structures did not increase the mobile or non-mobile invertebrate species richness on structure surfaces.

POPULATION RESPONSE (3 STUDIES)

- **Invertebrate abundance (3 studies):** Three randomized, controlled studies (including two replicated and one paired sites study) in Australia[1,2] and France[3] found that creating short flexible habitats on subtidal artificial structures had mixed effects on the mobile[1,2] and/or non-mobile[2,3] invertebrate abundance on and around structure surfaces, depending on the survey week[1], species group[1,2], flexible habitat length[2], or site[3]. One of the studies[3] found no effect on mobile invertebrate abundance.
- **Fish abundance (1 study):** One randomized, controlled study in Australia[1] found that creating short flexible habitats on subtidal artificial structures had mixed effects on the seahorse abundance on and around structures, depending on the survey week.

BEHAVIOUR (0 STUDIES)

Background

Short flexible habitats, such as understory macroalgal blades, turfs and soft-bodied invertebrates, provide other organisms three-dimensional habitat space and refuge from predation in subtidal rocky habitats (Levin & Hay 1996). They can support high biodiversity (Smale *et al.* 2020) but can also dominate space and have negative effects on other species (O'Brien & Scheibling 2018). The size, density and material properties of flexible habitats are likely to affect the size, abundance and variety of organisms that can use them and the spaces they create.

Some organisms that form flexible habitats tend to be absent or sparse on artificial subtidal structures (Wilhelmsson & Malm 2008), although some readily colonize in suitable conditions. Artificial flexible habitats such as ropes or nets can be present on some structures, but are likely to be temporary and regularly disturbed (e.g. removed and replaced) when present. Short flexible habitats can be created on subtidal artificial structures by adding material, either during construction or retrospectively. Material choice is important for creating flexible habitats, since some flexible materials are unlikely to persist in the marine environment, while those that do may become entanglement hazards or contribute to pollution if dislodged.

There is a body of literature describing the use of artificial turfs as collectors to measure larval supply and settlement in subtidal rocky habitats and to investigate the effects of structural complexity on ecological interactions (e.g. Atilla & Fleeger 2000; Perrett *et al.* 2006; Underwood & Chapman 2006). These studies are not included in this synopsis, which focusses on *in situ* conservation actions to enhance the biodiversity of structures that are engineered to fulfil a primary function other than providing artificial habitats. Studies that investigate the effects of transplanting live soft-bodied organisms onto structures are not included here, but are considered under the action "*Transplant or seed organisms onto subtidal artificial structures*".

Definition: 'Short flexible habitats' are flexible protruding materials such as rope, ribbon or twine 1–50 mm in length (modified from "Soft structures" in Strain *et al.* 2018).

See also: *Create long flexible habitats (>50 mm) on subtidal artificial structures*; *Transplant or seed organisms onto subtidal artificial structures*.

Atilla N. & Fleeger J.W. (2000) Meiofaunal colonization of artificial substrates in an estuarine embayment. *Marine Ecology*, 21, 69–83.

Levin P.S. & Hay M.E. (1996) Responses of temperate reef fishes to alterations in algal structure and species composition. *Marine Ecology Progress Series*, 134, 37–47.

O'Brien J.M. & Scheibling R.E. (2018) Turf wars: competition between foundation and turf-forming species on temperate and tropical reefs and its role in regime shifts. *Marine Ecology Progress Series*, 590, 1–17.

Perrett L.A., Johnston E.L. & Poore A.G.B. (2006) Impact by association: direct and indirect effects of copper exposure on mobile invertebrate fauna. *Marine Ecology Progress Series*, 326, 195–205.

Smale D.A., Epstein G., Hughes E., Mogg A.O.M. & Moore P.J. (2020) Patterns and drivers of understory macroalgal assemblage structure within subtidal kelp forests. *Biodiversity and Conservation*, 29, 4173–4192.

Strain E.M.A., Olabarria C., Mayer-Pinto M., Cumbo V., Morris R.L., Bugnot A.B., Dafforn K.A., Heery E., Firth L.B., Brooks P.R. & Bishop M.J. (2018) Eco-engineering urban infrastructure for marine and coastal biodiversity: which interventions have the greatest ecological benefit? *Journal of Applied Ecology*, 55, 426–441.

Underwood A.J. & Chapman M.G. (2006) Early development of subtidal macrofaunal assemblages: relationships to period and timing of colonization. *Journal of Experimental Marine Biology and Ecology*, 330, 221–233.

Wilhelmsson D. & Malm T. (2008) Fouling assemblages on offshore wind power plants and adjacent substrata. *Estuarine, Coastal and Shelf Science*, 79, 459–466.

A randomized, controlled study in 2008 on two subtidal swimming-enclosure nets in Sydney Harbour estuary, Australia (1) found that creating short flexible habitats (frayed-netting) on enclosure-net panels had mixed effects on seahorse *Hippocampus whitei* and mobile invertebrate abundances, depending on the survey week and invertebrate species group. Over two months, net panels with frayed-netting had higher seahorse abundance (1–3 individuals/panel) than panels without flexible habitats (0–1/panel) during six of seven surveys, but similar abundance during the other survey (frayed-netting: 1/panel; without: 0/panel). Mobile invertebrate abundances on panels with and without flexible habitats varied depending on the species group and survey week (see paper for results). Short flexible habitats were created on polyethylene rope swimming-enclosure nets (100 mm mesh size) in March 2008 by attaching clumps of frayed nylon netting (50 mm length) at knot intersections ('frayed-netting'). Three net panels (length: 0.3 m, height: 1 m) with frayed-netting and three panels without were randomly arranged along each of two enclosure nets (depth not reported). In May 2008, sixty-three seahorses were released onto the nets. Seahorses were counted on panels with and without flexible habitats over two months and mobile invertebrates (seahorse prey) were surveyed using a suction-pump over three months.

A replicated, randomized, controlled study in 2012–2013 on a subtidal dock in Sydney Harbour estuary, Australia (2) found that creating short flexible habitats (polyethylene turf) on settlement plates altered the non-mobile invertebrate community composition on plates and had mixed effects on the mobile invertebrate community composition and invertebrate abundances, depending on the turf length and species group. After three months, non-mobile invertebrate community composition differed on settlement plates with longer and shorter turf, and both differed to plates without turf (data reported as statistical model results). Plates with longer turf also supported different mobile invertebrate composition to plates with shorter turf and without turf, which were similar. Non-mobile invertebrates were less abundant on plates with turf (0–7% cover) than without (4–28%) in nine of 14 comparisons, but similar in the other five comparisons (with turf: 5–25%; without: 4–28%). Mobile invertebrates were more abundant on plates with turf (2–324 individuals/plate) than without (0–50/plate) in 22 of 28 comparisons, but similar in six comparisons (with turf: 2–58/plate; without: 1–50/plate). Plastic settlement plates (100 × 100 mm) were made with and without short flexible habitats (polyethylene turf). Plates with turf had either longer (18 mm) or shorter (2–3 mm) blades (1.5 mm width). Twelve of each were randomly arranged at 3 m depth beneath a dock with turf facing downwards in October 2012. Invertebrates on plates were counted in the laboratory after three months.

A replicated, randomized, paired sites, controlled study in 2014 on eight subtidal pontoons in two marinas in the English Channel and the Élorn estuary, France (3) found that creating short flexible habitats (polypropylene turf) on settlement plates did not increase the invertebrate species richness or the mobile invertebrate abundance on plates, but had mixed effects on the non-mobile invertebrate abundance and the community composition, depending on the turf density and site. Mobile invertebrate species richness and abundance was similar on plates with high-density turf (22–33 species/plate, 189–1,093 individuals/plate), low-density turf (23–34 species/plate, 194–1,132 individuals/plate) and plates without turf (19–27 species/plate, 132–1,019 individuals/plate). The same was true for non-mobile invertebrate species richness (high-density: 6–10 species/plate; low-density: 8–11/plate; no turf: 7–12/plate), and their abundance at one of two sites (high-density: 95–143% cover; low-density: 90–114%; no turf: 101–119%). At the second site, abundance was lower on plates with turf (high-density: 108–156%; low-density: 117–151%) than without (120–192%). Invertebrate community composition differed on plates with and without turf in four of eight comparisons, but was similar in the other four (data reported as statistical model results). Plastic settlement plates (180 × 180 mm) were made with and without short flexible habitats (polypropylene turf). Plates with turf (blade length: 30 mm; width: 2 mm) had either high (100% cover) or low (50%) turf density. One of each was randomly arranged vertically at 1 m depth beneath each of four pontoons in each of two marinas in May 2014. Invertebrates on plates were counted in the laboratory after three months.

(1) Hellyer C.B., Harasti D. & Poore A.G.B. (2011) Manipulating artificial habitats to benefit seahorses in Sydney Harbour, Australia. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 21, 582–589.
(2) Lavender J.T., Dafforn K.A., Bishop M.J. & Johnston E.L. (2017) Small-scale habitat complexity of artificial turf influences the development of associated invertebrate assemblages. *Journal of Experimental Marine Biology and Ecology*, 492, 105–112.
(3) Leclerc J.-C. & Viard F. (2018) Habitat formation prevails over predation in influencing fouling communities. *Ecology and Evolution*, 8, 477–492.

3.16. Create long flexible habitats (>50 mm) on subtidal artificial structures

- **Five studies** examined the effects of creating long flexible habitats on subtidal artificial structures on the biodiversity of those structures. Three studies were in estuaries in southeast Australia[1,2,3] and two were in a port in the Netherlands[4a,b].

COMMUNITY RESPONSE (2 STUDIES)

- **Overall community composition (2 studies):** Two replicated, controlled studies (including one randomized study) in Australia[1] and the Netherlands[4a] reported that long flexible habitats created on subtidal artificial structures supported macroalgae and non-mobile invertebrate[4a] or fish[1] species that were absent from on and around structure surfaces without flexible habitats.
- **Invertebrate community composition (1 study):** One replicated, controlled study in the Netherlands[4a] reported that creating long flexible habitats on subtidal artificial structures altered the non-mobile invertebrate community composition on structure surfaces[4a].
- **Fish richness/diversity (1 study):** One replicated, randomized, controlled study in Australia[1] found that creating long flexible habitats on subtidal artificial structures had mixed effects on the

fish species richness around structures, depending on fish presence when flexible habitats were created.

POPULATION RESPONSE (4 STUDIES)

- **Overall abundance (1 study):** One replicated, controlled study in the Netherlands[4a] reported that long flexible habitats created on subtidal artificial structures supported higher combined macroalgae and invertebrate (mostly mussels) biomass than structure surfaces without flexible habitats, and found that deeper flexible habitats supported higher biomass than shallower ones.
- **Invertebrate abundance (3 studies):** Two of three studies (including two replicated, two controlled and one randomized study) in Australia[3] and the Netherlands[4a,4b] found that creating long flexible habitats on subtidal artificial structures had mixed effects on the mobile[3,4b] and/or non-mobile[4b] invertebrate abundance on and around structure surfaces, depending on the species group and survey week[3], or the flexible habitat length and density[4b]. One study[4a] reported that creating flexible habitats decreased the mussel abundance on structure surfaces but that the flexible habitats themselves supported higher biomass (mostly mussels) than the structure surfaces.
- **Fish abundance (2 studies):** Two randomized, controlled studies (including one replicated study) in Australia[1,3] found that creating long flexible habitats on subtidal artificial structures had mixed effects on the abundance of fishes[1] or seahorses[3] on and around structures, depending on the species and fish presence when flexible habitats were created[1], or the survey week[3].

BEHAVIOUR (1 STUDY)

- **Use (1 study):** One replicated study in Australia[2] reported that long flexible habitats created on subtidal artificial structures were used by seahorses.

Background

Long flexible habitats, such as macroalgal canopies and soft-bodied invertebrates, provide other organisms three-dimensional habitat space and refuge from predation in subtidal rocky habitats (Levin & Hay 1996; Smale *et al.* 2020). The size, density and material properties of flexible habitats are likely to affect the size, abundance and variety of organisms that can use them and the spaces they create.

Some organisms that form flexible habitats tend to be absent or sparse on artificial subtidal structures (Wilhelmsson & Malm 2008), although some readily colonize in suitable conditions. Artificial flexible habitats such as ropes or nets can be present on some structures, but are likely to be temporary and regularly disturbed (e.g. removed and replaced) when present. Long flexible habitats can be created on subtidal artificial structures by adding material, either during construction or retrospectively. Material choice is important for creating flexible habitats, since some flexible materials are unlikely to persist in the marine environment, while those that do may become entanglement hazards or contribute to pollution if dislodged. Studies that investigate the effects of transplanting live soft-bodied organisms onto structures are not included here, but are considered under the action "*Transplant or seed organisms onto subtidal artificial structures*".

There are bodies of literature describing the use of artificial flexible habitats to investigate the effects of structural complexity on ecological interactions in subtidal rocky habitats (e.g. Shelamoff *et al.* 2020), for artificial reefs or fish aggregation devices (e.g. Kellison & Sedberry 1998; Vega Fernández *et al.* 2009), and for bivalve or seaweed cultivation (e.g. Peteiro *et al.* 2007; Walls *et al.* 2019). There are also laboratory-based studies investigating species preferences for different flexible habitats (e.g. Hellyer *et al.* 2011). These studies are not included in this synopsis, which focusses on *in situ* conservation actions to enhance the biodiversity of structures that are engineered to fulfil a primary function other than providing artificial habitats.

Definition: 'Long flexible habitats' are flexible protruding materials such as rope, ribbon or twine >50 mm in length (modified from "Soft structures" in Strain *et al.* 2018).

See also: *Create short flexible habitats (1–50 mm) on subtidal artificial structures*; *Transplant or seed organisms onto subtidal artificial structures*.

Hellyer C.B., Harasti D. & Poore A.G.B. (2011) Manipulating artificial habitats to benefit seahorses in Sydney Harbour, Australia. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 21, 582–589.

Kellison G.T. & Sedberry G.R. (1998) The effects of artificial reef vertical profile and hole diameter on fishes off South Carolina. *Bulletin of Marine Science*, 62, 763–780.

Levin P.S. & Hay M.E. (1996) Responses of temperate reef fishes to alterations in algal structure and species composition. *Marine Ecology Progress Series*, 134, 37–47.

Peteiro L.G., Filgueira R., Labarta U. & Fernández-Reiriz M.J. (2007) Effect of submerged time of collector ropes on the settlement capacity of *Mytilus galloprovincialis* L. *Aquaculture Research*, 38, 1679–1681.

Shelamoff V., Layton C., Tatsumi M., Cameron M.J., Edgar G.J., Wright J.T. & Johnson C.R. (2020) Kelp patch size and density influence secondary productivity and diversity of epifauna. *Oikos*, 129, 331–345.

Smale D.A., Epstein G., Hughes E., Mogg A.O.M. & Moore P.J. (2020) Patterns and drivers of understory macroalgal assemblage structure within subtidal kelp forests. *Biodiversity and Conservation*, 29, 4173–4192.

Strain E.M.A., Olabarria C., Mayer-Pinto M., Cumbo V., Morris R.L., Bugnot A.B., Dafforn K.A., Heery E., Firth L.B., Brooks P.R. & Bishop M.J. (2018) Eco-engineering urban infrastructure for marine and coastal biodiversity: which interventions have the greatest ecological benefit? *Journal of Applied Ecology*, 55, 426–441.

Vega Fernández T., D'Anna G., Badalamenti F. & Pérez-Ruzafa A. (2009) Effect of simulated macroalgae on the fish assemblage associated with a temperate reef system. *Journal of Experimental Marine Biology and Ecology*, 376, 7–16.

Walls A.M., Edwards M.D., Firth L.B. & Johnson M.P. (2019) Ecological priming of artificial aquaculture structures: kelp farms as an example. *Journal of the Marine Biological Association of the United Kingdom*, 99, 729–740.

Wilhelmsson D. & Malm T. (2008) Fouling assemblages on offshore wind power plants and adjacent substrata. *Estuarine, Coastal and Shelf Science*, 79, 459–466.

One replicated, randomized, controlled study in 1989 on 36 subtidal pontoons in Port Hacking estuary, Australia (1) found that creating long flexible habitats (artificial seagrass units, ASUs) on pontoons did not increase the fish species richness or abundance under pontoons in one trial, but did in a second trial in which pontoons had been cleared of fishes initially. In the first trial, after six weeks, fish species richness and abundance (excluding blennies *Parablennius* sp.) was similar under pontoons with ASUs (3–4 species/pontoon, 5–7 individuals/pontoon) and those without (1–2 species/pontoon, 2–4 individuals/pontoon). In the second trial, six weeks after clearing fishes from beneath pontoons, species richness and abundance was higher under pontoons with ASUs (4–5 species/pontoon, 6–11 individuals/pontoon) than without (0–1 species and individuals/pontoon). Blenny abundance was similar under pontoons with and without ASUs (0–17 vs 0–22 individuals/pontoon) in both trials. Three species recorded under pontoons with ASUs were absent from those without. Long flexible habitats (ASUs) were created by suspending steel mesh sheets (7 m^2) with buoyant plastic fragments (length: 280 mm; density: 800/m^2) under pontoons. One ASU was attached at 0.3 m depth under each of six randomly-selected pontoons in each of three sites within an estuary in September 1989. Fishes under pontoons with ASUs and under six without were netted (1 mm mesh size) and counted after six weeks. The trial was repeated in October after clearing fishes from under pontoons. Five ASUs were dislodged and no longer provided habitat.

A replicated study in 2003–2004 on two subtidal jetties in Sydney Harbour estuary, Australia (2) reported that long flexible habitats (nets) created on jetty pilings were used by two species of seahorse. Over 10 months, between one and three White's seahorses *Hippocampus whitei* were seen on nets attached to jetty pilings during three of five surveys at each of two sites. One big-belly seahorse *Hippocampus abdominalis* was seen during three of the surveys at one site. Two juvenile seahorses were seen on nets. Long flexible habitats were created by attaching five nets (length: 5 m; height: 3 m; material not reported) to wooden jetty pilings at each of two sites in May 2003. Nets were in contact with the seabed (depth not reported). Seahorses on nets were counted over 10 months.

A randomized, controlled study in 2008 on two subtidal swimming-enclosure nets in Sydney Harbour estuary, Australia (3) found that creating long flexible habitats (double-netting) on enclosure-net panels had mixed effects on seahorse *Hippocampus whitei* and mobile invertebrate abundances, depending on the survey week and invertebrate species group. Over two months, net panels with double-netting had higher seahorse abundance (1/panel) than panels without (0/panel) during two of seven surveys, but similar abundance in the other five (both 0–1/panel). Mobile invertebrate abundances on panels with and without double-netting varied depending on the species group and survey week (see paper for results). Long flexible habitats were created on polyethylene rope swimming-enclosure nets (100 mm mesh size) in March 2008 by attaching a second layer of enclosure netting ('double-netting'). Three net panels (length: 0.3 m, height: 1 m) with double-netting and three panels without were randomly arranged along each of two enclosure nets (depth not reported). In May 2008, sixty-three seahorses were released onto the nets. Seahorses were counted on panels with and without flexible habitats over two months and mobile invertebrates (seahorse prey) were surveyed using a suction-pump over three months.

One replicated, controlled study in 2009 on seven subtidal jetty pilings in the Port of Rotterdam, the Netherlands (4a) reported that creating long flexible habitats ('hulas') on pilings altered the non-mobile invertebrate community composition and reduced mussel *Mytilus edulis* cover on piling surfaces, but that hulas supported higher macroalgae and invertebrate biomass (mostly mussels) than piling surfaces without flexible habitats. Data were not statistically tested unless stated. After eight months, hula ropes supported mussels (60% cover), nine macroalgae and other non-mobile invertebrate species (0–2% cover), and five mobile invertebrate species groups (1–10 to >100 individuals/rope). Piling surfaces under hulas had 50% barnacle cover (*Amphibalanus improvisus*), while pilings without flexible habitats had 50% mussel and 14% barnacle cover. At least eight species (2 macroalgae, 6 non-mobile invertebrates) recorded on hulas were absent from piling surfaces without. Biomass was 44–113 kg/m^2 on hulas and 10 kg/m^2 on surfaces without. Biomass was statistically similar on ropes at 0.5 m depth (6 g/cm) and 1 m (7 g/cm), and higher on both than those at 0 m (3 g/cm). Long flexible habitats were created by attaching nylon rope skirts ('hulas') around pilings in March 2009. Three overlapping hulas with 167 ropes/hula (rope diameter: 6 mm; length: 550 mm; density: 167/m) were attached around each of five wooden and two steel pilings, cleared of organisms, with one hula at each of 0, 0.5 and 1 m depths. Hulas were compared with subtidal surfaces (200 × 200 mm) on seven additional wooden/steel pilings without hulas, cleared of organisms. Macroalgae and invertebrates on hula ropes and piling surfaces were counted and biomass (wet weight) measured in the laboratory over eight months.

One replicated study in 2009 on five subtidal pontoons in the Port of Rotterdam, the Netherlands (4b) found that long flexible habitats ('hulas') created under pontoons supported different invertebrate biomass depending on the rope length and density. After eight months, around hula edges, biomass of mussels *Mytilus edulis*, and other mobile and non-mobile invertebrates was higher on hulas with long ropes (17–19 g/cm) than mixed-length ropes (14–16 g/cm). In hula centres, biomass was similar on both designs (long ropes: 9–13 g/cm; mixed: 10–15 g/cm). Biomass was higher on hulas with low-density ropes (15 g/cm) than medium-density (12 g/cm), and higher on both than those with high-density (9 g/cm). Long flexible habitats were created by suspending plastic frames with nylon rope skirts ('hulas', 12 mm rope diameter) beneath five pontoons in March 2009. Two hulas (2.0 × 1.6 m, 208 ropes/hula) had different rope lengths (long: 1.5 m; mixed: 0.3–1.5 m), while three hulas (2.3 × 0.9 m, 1.5 m rope length) had different rope densities (high: 64 ropes/m^2; medium: 32/m^2; low: 16/m^2). Invertebrates on hula ropes were counted and biomass (wet weight) measured in the laboratory over eight months.

(1) Hair C.A. & Bell J.D. (1992) Effects of enhancing pontoons on abundance of fish: initial experiments in estuaries. *Bulletin of Marine Science*, 51, 30–36.

(2) Clynick B.G. (2008) Harbour swimming nets: a novel habitat for seahorses. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 18, 483–492.

(3) Hellyer C.B., Harasti D. & Poore A.G.B. (2011) Manipulating artificial habitats to benefit seahorses in Sydney Harbour, Australia. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 21, 582–589.

(4) Paalvast P., van Wesenbeeck B.K., van der Velde G. & de Vries M.B. (2012) Pole and pontoon hulas: an effective way of ecological engineering to increase productivity and biodiversity in the hard-substrate environment of the port of Rotterdam. *Ecological Engineering, 44, 199–209.*

3.17. Reduce the slope of subtidal artificial structures

- We found no studies that evaluated the effects of reducing the slope of subtidal artificial structures on the biodiversity of those structures.

This means we did not find any studies that directly evaluated this intervention during our literature searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Background

The slope of substrate surfaces can influence the species that colonize in subtidal rocky habitats (Harmelin-Vivien *et al*. 1995; Schroeter *et al.* 2015; but see Glasby 2000). Artificial structures tend to have steeper slopes than natural reefs, with narrower bands of subtidal habitat. This means that space for organisms is scarce and competitive interactions and other environmental processes differ (Chapman & Underwood 2011). Steep surfaces can also be associated with the presence of non-native species (Dafforn 2017).

Although fundamental aspects of structure designs, such as their slope, are likely to be driven by engineering and cost requirements, there may be opportunities to reduce the slope of subtidal artificial structure surfaces with the aim of enhancing their biodiversity. This may, however, lead to an increase in the physical footprint of structures and associated impacts on the receiving environment (Perkins *et al.* 2015). For this reason, studies that test the effects of creating additional artificial habitat in front of existing structures to create horizontal or gently sloping surfaces are not included in this synopsis, although such actions can deliver biodiversity benefits and are informative (e.g. Liversage & Chapman 2018; Toft *et al.* 2013).

There is a body of literature investigating the effects of slope on artificial reefs (e.g. Schroeter *et al.* 2015; Terawaki *et al.* 2003). These studies are not included in this synopsis, which focusses on *in situ* conservation actions to enhance the biodiversity of structures that are engineered to fulfil a primary function other than providing artificial habitats.

Definition: 'Reducing the slope' includes actions taken to reduce the inclination of structures without increasing the footprint, with the aim of enhancing their biodiversity.

See also: *Create small protrusions (1–50 mm) on subtidal artificial structures*; *Create large protrusions (>50 mm) on subtidal artificial structures*; *Create small ridges or ledges (1–50 mm) on subtidal artificial structures*; *Create large ridges or ledges (>50 mm) on subtidal artificial structures*.

Chapman M.G. & Underwood A.J. (2011) Evaluation of ecological engineering of "armoured" shorelines to improve their value as habitat. *Journal of Experimental Marine Biology and Ecology*, 400, 302–313.

Dafforn K.A. (2017) Eco-engineering and management strategies for marine infrastructures to reduce establishment and dispersal of non-indigenous species. *Management of Biological Invasions*, 8, 153–161.

Glasby T.M. (2000) Surface composition and orientation interact to affect subtidal epibiota. *Journal of Experimental Marine Biology and Ecology*, 248, 177–190.

Harmelin-Vivien M.L., Harmelin J.G. & Leboulleux V. (1995) Microhabitat requirements for settlement of juvenile sparid fishes on Mediterranean rocky shores. *Hydrobiologia*, 300, 309–320.

Liversage K. & Chapman M.G. (2018) Coastal ecological engineering and habitat restoration: incorporating biologically diverse boulder habitat. *Marine Ecology Progress Series*, 593, 173–185.

Perkins M.J., Ng T.P.T., Dudgeon D., Bonebrake T.C. & Leung K.M.Y. (2015) Conserving intertidal habitats: what is the potential of ecological engineering to mitigate impacts of coastal structures? *Estuarine, Coastal and Shelf Science*, 167, 504–515.

Schroeter S.C., Reed D.C. & Raimondi P.T. (2015) Effects of reef physical structure on development of benthic reef community: a large-scale artificial reef experiment. *Marine Ecology Progress Series*, 540, 43–55.

Terawaki T., Yoshikawa K., Yoshida G., Uchimura M. & Iseki K. (2003) Ecology and restoration techniques for *Sargassum* beds in the Seto Inland Sea, Japan. *Marine Pollution Bulletin*, 47, 1–6.

Toft J.D., Ogston A.S., Heerhartz S.M., Cordell J.R. & Flemer E.E. (2013) Ecological response and physical stability of habitat enhancements along an urban armored shoreline. *Ecological Engineering*, 57, 97–108.

3.18. Transplant or seed organisms onto subtidal artificial structures

- **Eleven studies** examined the effects of transplanting or seeding species onto subtidal artificial structures on the biodiversity of those structures. Eight studies were on open coastlines in Japan[1], Italy[4a,4b,4c,4d,5a,5b] and Croatia[5c], and one of each was in an inland bay in eastern USA[2], an estuary in southeast Australia[3], and on an island coastline in the Singapore Strait[6].

COMMUNITY RESPONSE (2 STUDIES)

- **Overall community composition (1 study):** One replicated, paired sites, controlled study in the USA[2] found that transplanting oysters onto subtidal artificial structures altered the combined invertebrate and fish community composition on and around structure surfaces.
- **Overall richness/diversity (1 study):** One replicated, paired sites, controlled study in the USA[2] found that transplanting oysters onto subtidal artificial structures increased the combined invertebrate and fish species richness and diversity on and around structure surfaces.
- **Invertebrate richness/diversity (1 study):** One randomized, before-and-after study in Singapore[6] reported that transplanting corals onto a subtidal artificial structure increased the coral species richness on structure surfaces.

POPULATION RESPONSE (11 STUDIES)

- **Overall abundance (1 study):** One replicated, paired sites, controlled study in the USA[2] found that transplanting oysters onto subtidal artificial structures did not increase the combined invertebrate and fish abundance on and around structure surfaces, but that the effects varied for different species.
- **Algal abundance (3 studies):** Two replicated, randomized, controlled studies in Italy[4d] and Croatia[5c] found that the cover of canopy algae transplanted onto subtidal artificial structures increased[5c] and/or was higher[4d,5c] when transplanted under cages but decreased[5c] and/or was lower[4d,5c] when left uncaged. One study in Japan[1] reported that the abundance of kelp recruits

on a subtidal artificial structure varied depending on the distance from transplanted kelp individuals and the surface orientation.

- **Invertebrate abundance (2 studies):** One replicated, randomized, controlled and site comparison study in Australia[3] found that transplanting sea urchins onto a subtidal artificial structure reduced the cover of non-native sea mat on kelps growing on the structure. One randomized, before-and-after study in Singapore[6] reported that transplanting corals increased the coral cover on structure surfaces.
- **Algal reproductive success (1 study):** One study in Japan[1] reported that kelp transplanted onto a subtidal artificial structure appeared to reproduce.
- **Invertebrate reproductive success (1 study):** One replicated, paired sites, controlled study in the USA[2] reported that oysters transplanted onto subtidal artificial structures appeared to reproduce.
- **Algal survival (5 studies):** Three of five replicated studies (including two randomized, controlled studies) in Italy[4a,4b,4c,4d,5b] found that the survival of canopy algae transplanted onto subtidal artificial structures varied depending on the wave-exposure and surrounding habitat[4a] or the presence[4d,5b] and/or mesh size[5b] of cages around transplants, while in one[4a] the surface orientation had no effect. Two studies[4b,4c] reported that no canopy algae transplants survived, and in one[4c] this was regardless of the presence of cages.
- **Invertebrate survival (3 studies):** One randomized, before-and-after study in Singapore[6] found that the survival of corals transplanted onto a subtidal artificial structure varied depending on the species. One replicated, paired sites, controlled study in the USA[2] found that cleaning activities did not affect survival of transplanted oysters. One replicated, randomized, controlled and site comparison study in Australia[3] simply reported that transplanted sea urchins survived.
- **Algal condition (3 studies):** Two replicated studies (including one randomized, controlled study) in Italy[4a,5a] found that the growth of canopy algae transplanted onto subtidal artificial structures varied depending on the wave-exposure and surface orientation[4a] or the presence of cages around transplants[5a], while in one[5a] the mesh size of cages had no effect. One study in Japan[1] simply reported that transplanted kelp grew.
- **Invertebrate condition (2 studies):** One randomized, before-and-after study in Singapore[6] reported that the growth of corals transplanted onto a subtidal artificial structure varied depending on the species. One replicated, paired sites, controlled study in the USA[2] reported that cleaning activities did not affect the growth of transplanted oysters.

BEHAVIOUR (0 STUDIES)

Background

Natural subtidal rocky habitats tend to support many more species than do artificial structures (Wilhelmsson & Malm 2008). Species can be transplanted or seeded directly onto structures with the aim of enhancing their biodiversity. The choice of species to transplant or seed depends on overarching management objectives. There may be value in transplanting rare, threatened, or commercially-valuable species to create artificial surrogate habitats for localized populations. Alternatively, keystone species such as primary producers, grazers, habitat-providers and water-quality regulators may be of interest to create productive and well-regulated artificial ecosystems. Species common to natural reefs may be the focus if, for example, larvae/spores cannot disperse far enough or hydrographic barriers (currents/tides) prevent recruitment, or to pre-empt potential non-native species invasions on new substrates.

When planning transplanting or seeding interventions, the method and timing of intervention and the suitability of the receiving environment should be carefully considered, as these factors may impact the likelihood of success. It is crucial to understand the reasons for a species' absence in the first place. If, for example, a species has declined due to poor water quality and there has been no intervening improvement, there can be little expectation of successful recovery following transplanting or seeding (Stevenson *et al.* 1993). Similarly, species that naturally occur in wave-sheltered habitats should not be expected to survive and thrive on wave-exposed structures (Jonsson *et al.* 2006). Biological factors such as predation pressure, competition and food availability may also affect outcomes and should be understood prior to carrying out the intervention (Gianni *et al.* 2018).

There are bodies of literature describing transplanting or seeding species into natural or artificial subtidal habitats to investigate ecological processes and interactions (e.g. Clynick *et al.* 2007; Marzinelli *et al.* 2009), for aquaculture (e.g. James *et al.* 2007; Walls *et al.* 2019), biocontrol (e.g. Atalah *et al.* 2016), restoration in natural habitats (e.g. Edwards *et al.* 2015; Vanderklift *et al.* 2020) and artificial reefs (e.g. Noordin Raja Omar *et al.* 1994; Shelamoff *et al.* 2020). These studies are not included in this synopsis, which focusses on *in situ* conservation actions to enhance the biodiversity of structures that are engineered to fulfil a primary function other than providing artificial habitats.

Definition: 'Transplanting or seeding species' includes actions taken to attach live organisms at any life stage onto structures, with the aim of generating self-sustaining populations.

Atalah J., Newcombe E.M. & Zaiko A. (2016) Biocontrol of fouling pests: effects of diversity, identity and density of control agents. *Marine Environmental Research*, 115, 20–27.

Clynick B.G., Chapman M.G. & Underwood A.J. (2007) Effects of epibiota on assemblages of fish associated with urban structures. *Marine Ecology Progress Series*, 332, 201–210.

Edwards A.J., Guest J.R., Heyward A.J., Villanueva R.D., Baria M.V., Bollozos I.S. & Golbuu Y. (2015) Direct seeding of mass-cultured coral larvae is not an effective option for reef rehabilitation. *Marine Ecology Progress Series*, 525, 105–116.

Gianni F., Bartolini F., Airoldi L. & Mangialajo L. (2018) Reduction of herbivorous fish pressure can facilitate focal algal species forestation on artificial structures. *Marine Environmental Research*, 138, 102–109.

James D.S., Day R.W. & Shepherd S.A. (2007) Experimental abalone ranching on artificial reef in Port Phillip Bay, Victoria. *Journal of Shellfish Research*, 26, 687–695.

Jonsson P.R., Granhag L., Moschella P.S., Åberg P., Hawkins S.J. & Thompson R.C. (2006) Interactions between wave action and grazing control the distribution of intertidal macroalgae. *Ecology*, 87, 1169–1178.

Marzinelli E.M., Zagal C.J., Chapman M.G. & Underwood A.J. (2009) Do modified habitats have direct or indirect effects on epifauna? *Ecology*, 90, 2948–2955.

Noordin Raja Omar R.M., Eng Kean C., Wagiman S., Mutalib Mat Hassan A., Hussein M., Bidin Raja Hassan R. & Omar Mat Hussin C. (1994) Design and construction of artificial reefs in Malaysia. *Bulletin of Marine Science*, 55, 1050–1061.

Shelamoff V., Layton C., Tatsumi M., Cameron M.J., Edgar G.J., Wright J.T. & Johnson C.R. (2020) Kelp patch size and density influence secondary productivity and diversity of epifauna. *Oikos*, 129, 331–345.

Stevenson J.C., Staver L.W. & Staver K.W. (1993) Water quality associated with survival of submersed aquatic vegetation along an estuarine gradient. *Estuaries*, 16, 346–361.

Venderklift M.A., Doropoulos C., Gorman D., Leal I., Minne A.J.P., Statton J., Steven A.D.L. & Wernberg T. (2020) Using propagules to restore coastal marine ecosystems. *Frontiers in Marine Science*, 7, 724.

Walls A.M., Edwards M.D., Firth L.B. & Johnson M.P. (2019) Ecological priming of artificial aquaculture structures: kelp farms as an example. *Journal of the Marine Biological Association of the United Kingdom*, 99, 729–740.

Wilhelmsson D. & Malm T. (2008) Fouling assemblages on offshore wind power plants and adjacent substrata. *Estuarine, Coastal and Shelf Science*, 79, 459–466.

A study in 2003–2005 on a subtidal breakwater on open coastline in Tosa Bay, Japan (1) reported that kelp *Ecklonia cava* transplanted onto concrete blocks placed on the breakwater grew and appeared to reproduce. Over 22 months, transplanted kelp grew to reproductive size (data not reported) and new recruits appeared on the surrounding breakwater surfaces. After 22 months, there were 0–53 kelp recruits/m^2 within 10 m of the transplants, depending on the distance from transplants and the orientation of the surface (data reported from Figure 4 in original paper). Recruits grew to 260–360 mm length. Kelp seedlings (100 mm length) were attached to ropes fixed on two concrete blocks and transplanted onto a concrete breakwater at 4 m depth in April 2003. Other details were reported in Japanese. Transplanted kelp were monitored and new recruits were counted on breakwater surfaces over 22 months.

A replicated, paired sites, controlled study in 2008 on eight subtidal pontoons in the Delaware Inland Bays, USA (2) found that 29–89% of oysters *Crassostrea virginica* transplanted onto floats attached to the pontoons survived and grew, regardless of cleaning frequency, and that transplanting oysters increased the invertebrate and fish species richness and diversity on and around floats, but had mixed effects on abundances, depending on the species. Over four months, transplanted oyster survival (29–89%) and growth (5–25 mm) was similar on floats cleaned every two or four weeks. In total, 23 mobile invertebrate and fish species were recorded on and around floats with transplanted oysters and 17 on and around floats without, while 11 non-mobile invertebrate species were recorded on transplanted oyster shells. Average mobile invertebrate and fish species diversity (reported as Simpson's and Evenness indices) and richness was higher on and around floats with transplanted oysters (8–10 species/float) than without (4–7/float), and their combined abundance was similar (data not reported), although abundances varied by species (see paper for results). Oysters supported on average seven non-mobile invertebrate species/float. Mobile invertebrate and fish community composition differed on and around floats with and without oysters (data reported as statistical model results). Oyster recruits were seen on transplanted oysters. Hatchery-reared oysters (61 mm average length) were transplanted into wire baskets (25 mm mesh size) submerged 0.2 m beneath plastic floats (1.0 × 0.7 × 0.3 m) and attached to pontoons. One float with oysters (6 l) and one without were attached to each of eight pontoons in June 2008. Floats were cleaned every two weeks on four pontoons and every four weeks on four. Oyster survival and growth was monitored, non-mobile invertebrates on oyster shells were counted, and mobile invertebrates and fishes on and around floats were netted (3 mm mesh size) and counted over four months.

A replicated, randomized, controlled and site comparison study in 2006–2007 on 20 subtidal jetty pilings in Sydney Harbour estuary, Australia (3) reported that 100% of sea urchins *Holopneustes purpurascens* transplanted onto pilings survived, and found that transplanting urchins reduced the non-native sea mat cover (mostly *Membranipora membranacea*) on kelp *Ecklonia cava* growing on the piings. After one month, all transplanted sea urchins remained on pilings. Non-native sea mat cover on kelp was lower on pilings with transplanted urchins (0–19% cover) than on pilings without (29–89%), and similar to kelp on natural reefs in one of two trials (pilings with urchins: 0–6%; natural reefs: 1%), but higher on pilings in the second trial (pilings with urchins: 17–19%; natural reefs: 2–3%). Sea urchins (>50 mm diameter) were collected from natural reefs and transplanted onto kelp growing on wooden jetty pilings (1.5 × 1.5 m surfaces) at 0–3 m depth, with five urchins/piling. Five pilings with urchins and five without were randomly arranged in each of two sites on a jetty in November 2006. Transplanted urchins were counted and non-native sea mat cover on kelp blades was measured from photographs after one month. Sea mat cover was also measured on kelp on nearby natural reefs. The experiment was repeated in April 2007.

A replicated study in 2008–2009 on four subtidal breakwaters on open coastline in the Adriatic Sea, Italy (4a) reported that 0–33% of canopy algae *Cystoseira barbata* transplanted onto the breakwaters survived, depending on the wave-exposure and surrounding habitat, and that survivors grew. Data were not statistically tested. After one week, no transplanted canopy algae survived on breakwaters on sandy shorelines. On rocky shorelines, after eight months, average survival was 33% on wave-sheltered sides of breakwaters and 9% on wave-exposed sides. Survival was 3–44% on horizontal surfaces and 9–27% on vertical surfaces. On average, wave-sheltered transplants grew to 120 mm and wave-exposed transplants to 90 mm. Average length was 130 mm on horizontal and 60 mm on vertical surfaces. Some transplants survived 12 months and appeared to reproduce (no data reported). Boulders with attached juvenile canopy algae (50 mm length) were collected from natural reefs, fragmented and transplanted onto boulder breakwaters using epoxy putty. Fragments were attached in 12 patches, with five individuals/patch, on both the wave-sheltered and wave-exposed sides of each of four breakwaters in June 2008 (depth not reported). Of the 12 patches in each setting, four were on each of: horizontal surfaces with adult canopy algae, horizontal surfaces without, and vertical surfaces without. Two of the breakwaters were on rocky shorelines and two were on sandy shorelines. Transplants were monitored over eight months.

A replicated study in 2009 on two subtidal breakwaters on open coastline in the Adriatic Sea, Italy (4b) reported that juvenile canopy algae *Cystoseira barbata* transplanted onto breakwaters did not survive. After three days, no transplants remained on either breakwater. Boulders (100 mm diameter) with attached juvenile canopy algae were collected from a natural reef and transplanted onto boulder breakwaters using epoxy putty. Four boulders with canopy algae (numbers not reported) were attached on horizontal surfaces on the wave-sheltered side of each of two breakwaters on sandy shoreline in June 2009 (depth not reported). Transplants were monitored over three days.

A replicated, randomized, controlled study in 2009 on two subtidal breakwaters on open coastline in the Adriatic Sea, Italy (4c) reported that juvenile canopy algae *Cystoseira barbata* transplanted onto breakwaters did not survive, regardless of whether they were transplanted under cages or left uncaged. After two days, no transplants remained on either breakwater. Boulders (100 mm diameter) with attached juvenile canopy algae were collected from a natural reef and transplanted onto boulder breakwaters using epoxy putty. Eight boulders with canopy algae were attached on horizontal surfaces on the wave-sheltered side of each of two breakwaters on sandy shoreline in June 2009 (depth not reported). Four randomly-selected boulders on each breakwater were protected from grazers by plastic cages (10 mm mesh size) and four were left uncaged. Transplants were monitored over two days.

A replicated, randomized, controlled study in 2009 on two subtidal breakwaters on open coastline in the Adriatic Sea, Italy (4d) reported that 50–100% of juvenile canopy algae *Cystoseira barbata* transplanted onto the breakwaters survived, and found that survival and cover was higher when algae was transplanted under cages than when left uncaged. After eight days, average survival and remaining cover of transplanted canopy algae was higher under cages (100% survival, 88% of original cover) compared with uncaged transplants (50% survival, 24% cover). Limestone settlement plates (100 × 100 mm) were attached to rocky seabed at 3 m depth in March 2009 and were colonized by juvenile canopy algae. In June 2009, plates were removed and transplanted onto boulder breakwaters using epoxy putty. Eight plates were attached on horizontal surfaces on the wave-sheltered side of each of two breakwaters on sandy shoreline (depth not reported). Four randomly-selected plates on each breakwater were protected from grazers by plastic cages (1 mm mesh size) and four were left uncaged. Transplants were monitored over eight days.

A replicated, randomized, controlled study in 2010 on a subtidal breakwater on open coastline in the Adriatic Sea, Italy (5a) reported that canopy algae *Cystoseira barbata* transplanted onto the breakwater under cages grew, but decreased in length when left uncaged. Over 13 days, canopy algae transplant growth was similar under large-mesh cages (131% of original length) and small-mesh cages (115%), but uncaged transplants decreased in length (18% of original length). Boulders with attached juvenile canopy algae were collected from natural reefs, fragmented and attached to limestone plates (100 × 100 mm) using epoxy putty, then transplanted onto a boulder breakwater. Fifteen plates with 5–6 individuals/plate were attached to horizontal surfaces on the wave-sheltered side of a breakwater on sandy shoreline in July 2010 (depth not reported). Five randomly-selected plates were protected from grazers by large-mesh plastic-coated wire cages (10 mm mesh size), five by small-mesh cages (1 mm) and five were left uncaged.

Transplants were monitored over 13 days. Three caged plates were missing and no longer retained transplants on the breakwater.

A replicated, randomized, controlled study in 2010 on three subtidal breakwaters on open coastline in the Adriatic Sea, Italy (5b) reported that 67–100% of canopy algae *Cystoseira barbata* transplanted onto the breakwaters survived, depending on the presence and mesh-size of cages around them. After 15 days, canopy algae transplant survival was higher under small-mesh cages (100%) than large-mesh cages (75%) and for uncaged transplants (67%), which were similar. Boulders with attached juvenile canopy algae were collected from natural reefs, fragmented and attached to limestone plates (100 × 100 mm) using epoxy putty, then transplanted onto boulder breakwaters. Fifteen plates with 5–6 individuals/plate were attached to horizontal surfaces on the wave-sheltered side of each of three breakwaters on sandy shorelines in August 2010 (depth not reported). On each breakwater, five randomly-selected plates were protected from grazers by large-mesh plastic-coated wire cages (10 mm mesh size with 60 × 70 mm openings), five by small-mesh cages (1 mm) and five were left uncaged. Transplants were monitored over 15 days. Four caged plates were missing and no longer retained transplants on the breakwater.

A replicated, randomized, controlled study in 2010–2011 on three subtidal breakwaters on open coastline in the Adriatic Sea, Croatia (5c) reported that the cover of canopy algae transplanted onto breakwaters under cages increased, but decreased when left uncaged, and found that cover of caged transplants was higher than uncaged transplants. After 12 months, canopy algae transplant cover was higher under cages than when left uncaged for both *Cystoseira barbata* (caged: 72%; uncaged: 8%) and *Cystoseira compressa* (caged: 79%; uncaged: 13%) canopy algae. Cover increased by 7–31% under cages, but decreased by 40–55% when left uncaged. Limestone settlement plates (100 × 100 mm) were attached to rocky seabed at 3–4 m depth in May 2010 and were colonized by two species of juvenile canopy algae (40–70% cover). In October 2010, plates were removed and transplanted onto boulder breakwaters using epoxy putty. Eight plates of each species were attached to horizontal surfaces on the wave-sheltered side of each of three breakwaters on rocky shorelines (depth not reported). On each breakwater, four randomly-selected plates of each species were protected from grazers by plastic-coated wire cages (10 mm mesh size) and four were left uncaged. Transplants were monitored from photographs over 12 months.

A randomized, before-and-after study in 2015–2016 on a subtidal seawall on an island coastline in the Singapore Strait, Singapore (6) reported that 58–100% of corals transplanted onto the seawall survived, depending on the species, that most survivors grew, and that transplanting corals increased the coral species richness and cover on the seawall. After six months, average transplant survival was lower for *Pocillopora damicornis* (58%) than for all other hard coral species (*Echinopora lamellosa*: 100%; *Hydnophora rigida*: 100%; *Merulina ampliata*: 91%; *Platygyra sinensis*: 97%; *Podabacia crustacea*: 92%). Surviving *M. ampliata* transplants had negative growth rates (-1 cm^2/month), while all other species had positive growth rates (*E. lamellosa*: 11

cm^2/month; *H. rigida*: 14 cm^2/month; *P. sinensis*: 4 cm^2/month; *P. damicornis*: 26 cm^2/month; *P. crustacea*: 4 cm^2/month). Coral species richness and cover on the seawall was higher (8 species, 21% cover) than before corals were transplanted (2 species, 3% cover). Corals were collected from natural reefs, fragmented and reared on nursery tables adjacent to a granite boulder seawall at 4 m depth for nine months, before being transplanted onto the seawall using epoxy putty. Fragments (diameter: 9–16 cm; area: 48–160 cm^2) of six hard coral species (36 fragments/species) were randomly arranged in four patches on the seawall at 3m depth during April–August 2015. Corals were counted on the seawall (20 × 3 m section) before and six months after transplants were attached. Transplants were monitored from photographs over six months.

(1) Serisawa Y., Imoto Z., Imoto Y. & Matsuyama-Serisawa K. (2007) Density and growth of *Ecklonia cava* that appeared around artificial reefs with seedlings off Usa in Tosa Bay, Japan. *Aquaculture Science*, 55, 47–53.

(2) Marenghi F.P. & Ozbay G. (2010) Floating oyster, *Crassostrea virginica* Gmelin 1791, aquaculture as habitat for fishes and macroinvertebrates in Delaware Inland Bays: the comparative value of oyster clusters and lose shell. *Journal of Shellfish Research*, 29, 889–904.

(3) Marzinelli E.M., Underwood A.J. & Coleman R.A. (2011) Modified habitats influence kelp epibiota via direct and indirect effects. *PLoS ONE*, 6, e21936.

(4) Perkol-Finkel S., Ferrario F., Nicotera V. & Airoldi L. (2012) Conservation challenges in urban seascapes: promoting the growth of threatened species on coastal infrastructures. *Journal of Applied Ecology*, 49, 1457–1466.

(5) Ferrario F., Iveša L., Jaklin A., Perkol-Finkel S. & Airoldi L. (2016) The overlooked role of biotic factors in controlling the ecological performance of artificial marine habitats. *Journal of Applied Ecology*, 53, 16–24.

(6) Toh T.C., Ng C.S.L., Loke H.X., Taira D., Toh K.B., Afiq-Rosli L., Du R.C.P., Cabaitan P., Sam S.Q., Kikuzawa Y.P., Chou L.M. & Song T. (2017) A cost-effective approach to enhance scleractinian diversity on artificial shorelines. *Ecological Engineering*, 99, 349–357.

3.19. Control or remove non-native or nuisance species on subtidal artificial structures

- We found no studies that evaluated the effects of controlling or removing non-native or nuisance species on subtidal artificial structures on the biodiversity of those structures.

This means we did not find any studies that directly evaluated this intervention during our literature searches. Therefore, we have no evidence to indicate whether or not the intervention has any desirable or harmful effects.

Background

Marine artificial structures often support non-native and nuisance species (Airoldi *et al.* 2015; Dafforn 2017), especially those built in urban areas with high vessel movement, disturbance from human activity and poor water quality (Airoldi & Bulleri 2011; Mineur *et al.* 2012). This can have negative effects on native marine biodiversity on and around structures, as well as on humans.

It may be possible to control or remove non-native or nuisance species on subtidal artificial structures to enhance their biodiversity, using mechanical, chemical or biological methods. However, careful consideration must be given to what constitutes a non-native or nuisance species in any given location and scenario, to warrant its control or removal. In this synopsis, species are considered non-native when considered-so in the original study. However, 'nuisance' species that are not also non-native only includes those that have a negative effect on native biodiversity (e.g. by dominating space or smothering) – *not* those that are only a nuisance to society (e.g. by creating slippery surfaces, overgrowing aquaculture species, or fouling infrastructure). Care must also be taken to avoid causing unintended harm to non-target organisms (Locke *et al.* 2009).

Studies investigating control/removal actions that are indiscriminate and simultaneously remove all biodiversity from structure surfaces (e.g. Novak *et al.* 2017) or aim to prevent or reduce colonization in the first place for the benefit of humans (i.e. biofouling reduction; Scardino & de Nys 2011) are not included in this synopsis, which focusses on actions to enhance the biodiversity of artificial structures. Studies that only report the effects of actions on the controlled/removed species itself and not on the wider native biodiversity of structures are not included. Studies that report the effects of patch-scale control/removal, where continued presence on surrounding surfaces would be expected to influence conservation outcomes in practice, are not included but are informative (e.g. Dumont *et al.* 2011). Studies that investigate the effects of actions associated with maintenance or harvesting activities that reduce the likelihood of non-native species occupying bare space made available on structure surfaces following these activities are not considered here, but are included under "*Cease or alter maintenance activities on subtidal artificial structures*" and "*Manage or restrict harvesting of species on subtidal artificial structures*".

Definition: 'Controlling or removing non-native or nuisance species' includes actions taken to reduce the abundance of non-native or nuisance organisms on structures, with the aim of enhancing their biodiversity.

See also: *Cease or alter maintenance activities on subtidal artificial structures*; *Manage or restrict harvesting of species on subtidal artificial structures*.

Airoldi L. & Bulleri F. (2011) Anthropogenic disturbance can determine the magnitude of opportunistic species responses on marine urban infrastructures. *PLoS ONE*, 6, e22985.

Airoldi L., Turon X., Perkol-Finkel S. & Rius M. (2015) Corridors for aliens but not for natives: effects of marine urban sprawl at a regional scale. *Diversity and Distributions*, 21, 755–768.

Dafforn K.A. (2017) Eco-engineering and management strategies for marine infrastructures to reduce establishment and dispersal of non-indigenous species. *Management of Biological Invasions*, 8, 153–161.

Dumont C.P., Harris L.G. & Gaymer C.F. (2011) Anthropogenic structures as a spatial refuge from predation for the invasive bryozoan *Bugula neritina*. *Marine Ecology Progress Series*, 427, 95–103.

Locke A., Doe K.G., Fairchild W.L., Jackman P.M. & Reese E.J. (2009) Preliminary evaluation of effects of invasive tunicate management with acetic acid and calcium hydroxide on non-target marine organisms in Prince Edward Island, Canada. *Aquatic Invasions*, 4, 221–236.

Mineur F., Cook E.J., Minchin D., Bohn K., Macleod A. & Maggs C.A. (2012) Changing coasts: marine aliens and artificial structures. *Oceanography and Marine Biology: An Annual Review*, 50, 189–234.

Novak L., López-Legentil S., Sieradzki E. & Shenkar N. (2017) Rapid establishment of the non-indigenous ascidian *Styela plicata* and its associated bacteria in marinas and fishing harbors along the Mediterranean coast of Israel. *Mediterranean Marine Science*, 18, 324–331.
Scardino A.J. & de Nys R. (2011) Mini review: biomimetic models and bioinspired surfaces for fouling control. *Biofouling: The Journal of Bioadhesion and Biofilm Research*, 27, 73–86.

3.20. Cease or alter maintenance activities on subtidal artificial structures

- **Two studies** examined the effects of ceasing or altering maintenance activities on subtidal artificial structures on the biodiversity of those structures. One study was in an estuary in southeast Australia[1] and one was in an inland bay in eastern USA[2].

COMMUNITY RESPONSE (1 STUDY)

- **Overall community composition (1 study):** One replicated, paired sites, controlled study in the USA[2] found that reducing the frequency of cleaning on subtidal artificial structures did not alter the combined invertebrate and fish community composition on and around structure surfaces.
- **Overall richness/diversity (1 study):** One replicated, paired sites, controlled study in the USA[2] found that reducing the frequency of cleaning on subtidal artificial structures did not increase the combined invertebrate and fish species richness or diversity on and around structure surfaces.

POPULATION RESPONSE (2 STUDIES)

- **Overall abundance (1 study):** One replicated, paired sites, controlled study in the USA[2] found that reducing the frequency of cleaning on subtidal artificial structures did not increase the combined invertebrate and fish abundance on and around structure surfaces.
- **Algal abundance (1 study):** One replicated, paired sites, controlled study in the USA[2] found that reducing the frequency of cleaning on subtidal artificial structures increased the macroalgal abundance on structure surfaces.
- **Fish abundance (1 study):** One replicated, randomized, controlled study in Australia[1] found that reducing the area cleaned on a subtidal artificial structure increased the seahorse abundance on structure surfaces.
- **Survival (1 study):** One replicated, paired sites, controlled study in the USA[2] found that reducing the frequency of cleaning on subtidal artificial structures did not increase the survival of transplanted oysters.
- **Condition (1 study):** One replicated, paired sites, controlled study in the USA[2] found that reducing the frequency of cleaning on subtidal artificial structures did not increase the growth of transplanted oysters.

BEHAVIOUR (0 STUDIES)

Background

Subtidal rocky habitats experience intermittent disturbance from storms, sedimentation, pollution and other human activities, which lead to fluctuations in biodiversity (e.g.

Balata *et al.* 2007). These pressures are often more pronounced and frequent on artificial structures, especially those built in urban areas with high human activity and poor water quality, and/or in areas of high wave energy (Airoldi & Bulleri 2011; Moschella *et al.* 2005). Artificial structures are also often subject to disturbance from maintenance activities carried out to ensure they remain fit-for-purpose, safe, and aesthetically acceptable. Maintenance can include repairing or reinforcing points of weakness such as eroded cracks or holes, moving or replacing dislodged components, or cleaning regimes using physical or chemical methods. Such activities can further disturb, damage or remove biodiversity from structure surfaces (Harasti *et al.* 2010; Mamo *et al.* 2020), reduce the availability of microhabitats for organisms to shelter in (Moreira *et al.* 2007), and leave bare space available to opportunistic non-native or other nuisance species (Viola *et al.* 2017).

Although some maintenance is likely to be essential, there may be opportunities to cease or alter activities that disturb, damage or remove native organisms from subtidal artificial structures, to maintain or enhance their biodiversity. Altering activities could include using lower-impact methods, reducing the frequency or adjusting the timing of maintenance to avoid disturbance, damage, removal or the creation of bare space on surfaces when non-native or problematic species are more likely to occupy it (Airoldi & Bulleri 2011; Viola *et al.* 2017). It could also include allowing natural weathering of structure surfaces to occur, creating texture and microhabitat spaces (Moreira *et al.* 2007).

Studies of the effects of real or simulated maintenance activities to illustrate their impact compared with no or altered maintenance, where it is not clear that ceasing/altering maintenance would be a feasible conservation action, are not included but are informative (e.g. Airoldi & Bulleri 2011; Mamo *et al.* 2020; Viola *et al.* 2017). Studies that investigate cleaning activities to control or remove non-native or nuisance species are similarly not included where these actions are indiscriminate, simultaneously removing all biodiversity (e.g. Novak *et al.* 2017).

Definition: 'Ceasing or altering maintenance activities' includes actions taken to avoid or reduce the disturbance, damage or removal of native organisms from structures, with the aim of enhancing their biodiversity.

See also: *Control or remove non-native or nuisance species on subtidal artificial structures*; *Manage or restrict harvesting of species on subtidal artificial structures*.

Airoldi L. & Bulleri F. (2011) Anthropogenic disturbance can determine the magnitude of opportunistic species responses on marine urban infrastructures. *PLoS ONE*, 6, e22985.

Balata D., Piazzi L. & Benedetti-Cecchi L. (2007) Sediment disturbance and loss of beta diversity on subtidal rocky reefs. *Ecology*, 88, 2455–2461.

Harasti D., Glasby T.M. & Martin-Smith K.M. (2010) Striking a balance between retaining populations of protected seahorses and maintaining swimming nets. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 20, 159–166.

Mamo L.T., Porter A.G., Tagliafico A., Coleman M.A., Smith S.D.A., Figueira W.F. & Kelaher B.P. (2020) Upgrades of coastal protective infrastructure affect benthic communities. *Journal of Applied Ecology*, 58, 295–303.

Moreira J., Chapman M.G. & Underwood A.J. (2007) Maintenance of chitons on seawalls using crevices on sandstone blocks as habitat in Sydney Harbour, Australia. *Journal of Experimental Marine Biology and Ecology*, 347, 134–143.

Moschella P.S., Abbiati M., Åberg P., Airoldi L., Anderson J.M., Bacchiocchi F., Bulleri F., Dinesen G.E., Frost M., Gacia E., Granhag L., Jonsson P.R., Satta M.P., Sundelöf A., Thompson R.C. & Hawkins S.J. (2005) Low-crested coastal defence structures as artificial habitats for marine life: using ecological criteria in design. *Coastal Engineering*, 52, 1053–1071.

Novak L., López-Legentil S., Sieradzki E. & Shenkar N. (2017) Rapid establishment of the non-indigenous ascidian *Styela plicata* and its associated bacteria in marinas and fishing harbors along the Mediterranean coast of Israel. *Mediterranean Marine Science*, 18, 324–331.

Viola S.M., Page H.M., Zaleski S.F., Miller R.J., Doheny B., Dugan J.E., Schroeder D.M. & Schroeter S.C. (2017) Anthropogenic disturbance facilitates a non-native species on offshore oil platforms. *Journal of Applied Ecology*, 55, 1583–1593.

A replicated, randomized, controlled study in 2007–2008 on a subtidal swimming-enclosure net in Sydney Harbour estuary, Australia (1) found that enclosure-net panels cleaned only along the top section supported a higher abundance of seahorses *Hippocampus abdominalis* and *Hippocampus whitei* than panels cleaned only along the bottom or from top-to-bottom. Over four months, enclosure-net panels cleaned only along the top supported more seahorses (20% of original abundance) than panels cleaned along the bottom (5%) or from top-to-bottom (3%). Maintenance activities were altered on a polypropylene swimming-enclosure net (length: 150 m; height: 3–4 m from sea surface to seabed; mesh size: 100 mm) in November 2007. Net panels (4-m sections) were either cleaned along the top only (surface to 1 m depth), the bottom only (seabed to 1 m above), or from top-to-bottom (surface to seabed). There were four panels of each treatment, randomly arranged along the net. Seahorses were removed during cleaning then replaced in the same position, while all other organisms were scraped from the net. Some panels were left uncleaned but this treatment was not considered a feasible conservation action since the weight of the net could cause it to break or sink. Eleven seahorses on each panel were tagged and monitored over four months.

A replicated, paired sites, controlled study in 2008 on eight subtidal pontoons in the Delaware Inland Bays, USA (2) found that reducing the frequency of cleaning activity did not increase the survival or growth of transplanted oysters *Crassostrea virginica* on floats attached to the pontoons, nor did it alter the non-mobile invertebrate, mobile invertebrate and fish community composition or increase their species diversity, richness or abundance on and around floats, but it did increase the macroalgal abundance. Data for all comparisons were reported as statistical model results. Over four months, transplanted oyster survival and growth was similar on floats cleaned every four or two weeks. The same was true for the overall community composition, and the species diversity, richness and abundance of non-mobile invertebrates and of mobile invertebrates and fishes on and around oyster floats. The abundance of macroalgae was higher on floats cleaned every four than every two weeks. Maintenance activities were altered on floats holding transplanted oysters attached to pontoons during June–

September 2008. Hatchery-reared oysters (61 mm average length) were transplanted into wire baskets (25 mm mesh size) submerged 0.2 m beneath plastic floats (1.0 × 0.7 × 0.3 m) and attached to pontoons. One float with oysters (6 l) and one without were attached to each of eight pontoons in June 2008. Floats were cleaned with a freshwater hose every four weeks on four pontoons and every two weeks on four. Oyster survival and growth was monitored, non-mobile invertebrates on oyster shells were counted, and mobile invertebrates and fishes on and around floats were netted (3 mm mesh size) and counted over four months.

(1) Harasti D., Glasby T.M. & Martin-Smith K.M. (2010) Striking a balance between retaining populations of protected seahorses and maintaining swimming nets. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 20, 159–166.
(2) Marenghi F.P. & Ozbay G. (2010) Floating oyster, *Crassostrea virginica* Gmelin 1791, aquaculture as habitat for fishes and macroinvertebrates in Delaware Inland Bays: the comparative value of oyster clusters and lose shell. *Journal of Shellfish Research*, 29, 889–904.

3.21. Manage or restrict harvesting of species on subtidal artificial structures

- **Three studies** examined the effects of managing or restricting harvesting of species on subtidal artificial structures on the biodiversity of those structures or on human behaviour likely to influence the biodiversity of those structures. The studies were on open coastlines in Italy[1,2,3].

COMMUNITY RESPONSE (1 STUDY)

- **Fish community composition (1 study):** One site comparison study in Italy[3] found different fish community composition around subtidal artificial structures with and without harvesting restrictions. The structure with harvesting restrictions supported species that were absent from unrestricted structures.
- **Fish richness/diversity (1 study):** One site comparison study in Italy[3] found higher fish species richness around a subtidal artificial structure with harvesting restrictions compared with unrestricted structures.

POPULATION RESPONSE (2 STUDIES)

- **Invertebrate abundance (1 study):** One site comparison study in Italy[2] found similar sea urchin abundances around subtidal artificial structures with and without harvesting restrictions.
- **Fish abundance (2 studies):** One of two site comparison studies in Italy[2,3] found similar total fish abundance around subtidal artificial structures with and without harvesting restrictions, but that abundances varied depending on the species and the survey date[3]. One study[2] found higher seabream abundances around the structure with harvesting restrictions.

BEHAVIOUR (1 STUDY)

- **Human behaviour change (1 study):** One replicated, randomized study in Italy[1] reported that legally restricting human access on subidal artificial structures did not prevent people from harvesting invertebrates and fishes on and around structures.

Background

Subtidal rocky habitats experience intermittent disturbance from storms, sedimentation, pollution and other human activities, which lead to fluctuations in biodiversity (e.g. Balata *et al.* 2007). These pressures are often more pronounced and frequent on artificial structures, especially those built in urban areas with high human activity and poor water quality, and/or in areas of high wave energy (Airoldi & Bulleri 2011; Moschella *et al.* 2005). Artificial structures can also be subject to disturbance from recreational or commercial harvesting of species for food, bait, recreation or souvenirs. Such activities can further disturb, damage or remove biodiversity from structure surfaces (Bulleri & Airoldi 2005; Airoldi *et al.* 2005) and can leave bare space available to opportunistic non-native or other nuisance species (Viola *et al.* 2017).

In some circumstances, it may be desirable for structures to support multifunctional recreational or commercial activities, potentially diverting pressure away from natural habitats (Evans *et al.* 2017). If this is not the case, there may be opportunities to manage or restrict harvesting activities that disturb, damage or remove native organisms from subtidal artificial structures, to maintain or enhance their biodiversity. This could include introducing voluntary or enforced spatial or temporal restrictions, promoting sustainable alternatives, educating harvesters on the impacts of their activities, or a variety of other actions that may alter harvesting behaviour with the aim of enhancing the biodiversity of structures. Some artificial structures already have restricted access with existing surveillance. These may offer a means of creating cost-effective "artificial micro-reserves" where historical cultural rights preclude restricting activities in natural habitats (García-Gómez *et al.* 2010).

Studies of the effects of real or simulated harvesting to illustrate its impact compared with no or managed harvesting, where it is not clear that restricting/managing harvesting would be a feasible conservation action, are not included but are informative (e.g. Airoldi *et al.* 2005; Bulleri & Airoldi 2005).

Definition: 'Managing or restricting harvesting of species' includes actions taken to avoid or reduce the disturbance, damage or removal of native organisms from structures, with the aim of enhancing their biodiversity.

See also: *Control or remove non-native or nuisance species on subtidal artificial structures*; *Cease or alter maintenance activities on subtidal artificial structures*.

Airoldi L., Bacchiocchi F., Cagliola C., Bulleri F. & Abbiati M. (2005) Impact of recreational harvesting on assemblages in artificial rocky habitats. *Marine Ecology Progress Series*, 299, 55–66.
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A replicated, randomized study in 2001–2002 on subtidal breakwaters and groynes in five sites on open coastline in the Adriatic Sea, Italy (1) reported that making access to the breakwaters illegal did not prevent people from harvesting invertebrates and fishes on and around them. At four sites, an average of 0–2 harvesters/2-hour survey were recorded on breakwaters, despite access being illegal. At one site where breakwaters (access illegal) and groynes (access legal) were studied simultaneously, an average of 0–5 harvesters/2-hour survey were recorded. At this site >70% of observations were on groynes, but harvesting also occurred on breakwaters (details not reported). Harvesting species on breakwaters was restricted by making access illegal, but with no apparent enforcement (timing and other details not reported). The number of people harvesting invertebrates and fishes on breakwaters at each of five sites was counted during 2-hour surveys on 152 randomly-selected days between November 2001 and November 2002. Observations at one of the sites included harvesting on groynes, to which access was legal.

A site comparison study in 2002–2003 on three subtidal breakwaters on open coastline in the Adriatic Sea, Italy (2; same experimental set-up as 3) found higher abundances of white seabream *Diplodus sargus*, two-banded seabream *Diplodus vulgaris* and gilt-head seabream *Sparus aurata*, but similar abundance of sea urchins *Paracentrotus lividus* around a breakwater with restricted harvesting, compared with two unrestricted breakwaters. Sixteen years after harvesting restrictions were introduced, abundance was higher around the breakwater with restrictions than those without for white seabream in two of four surveys (restricted: 5–8 individuals/125m^2; unrestricted: 0–2/125m^2) and for two-banded seabream in three surveys (restricted: 2–46/125m^2; unrestricted: 0–14/125m^2). In the remaining surveys, abundances were similar around restricted (white: 3–10/125m^2; two-banded: 4/125m^2) and unrestricted breakwaters (white: 0–8/125m^2; two-banded: 1–3/125m^2). Gilt-head seabream were present only at the restricted breakwater in three of the surveys (1–2/125m^2) and was more abundant in the fourth (restricted: 2/125m^2; unrestricted: <1/125m^2). Urchin abundance was similar around restricted and unrestricted breakwaters (both 2–11/20m^2). Harvesting species on and around a boulder breakwater was restricted by creating a marine protected area in 1986, making fishing illegal with successful enforcement. Fishes and sea urchins were counted during four surveys at 4–7 m depth in 2002–2003 around the breakwater with restricted harvesting and around two nearby breakwaters with no restrictions.

A site comparison study in 2002–2003 on three subtidal breakwaters on open coastline in the Adriatic Sea, Italy (3; same experimental set-up as 2) found higher fish species richness and different fish community composition around a breakwater with restricted harvesting, compared with two unrestricted breakwaters, while fish abundances varied depending on the species and survey date. Sixteen years after harvesting restrictions were introduced, the fish species richness was higher around the breakwater with restrictions (24–27 species/breakwater) than those without (13–22/breakwater) and the fish community composition differed in seven of eight comparisons (data reported as statistical model results). Total fish abundance was higher around the restricted breakwater in only one of four surveys (152 vs 63–66 individuals/survey) but was similar in three (319–554 vs 192–841/survey). However, the individual abundances of eight of 12 fish species were higher around the restricted breakwater during two or more surveys (see paper for full results). Three fish species recorded around the restricted breakwater were absent from unrestricted breakwaters. Harvesting species on and around a boulder breakwater was restricted by creating a marine protected area in 1986, making fishing illegal with successful enforcement. Fishes were counted during four surveys at 4–7 m depth in 2002–2003 around the breakwater with restricted harvesting and around two nearby breakwaters with no restrictions.

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Appendix 1. Glossary of terms

Amphipods: Members of the phylum Crustacea. Mostly marine. Small invertebrate organisms resembling shrimps.

Ascidians: Also known as sea squirts or tunicates. Members of the phylum Chordata: subphylum Tunicata. Small sac-like marine invertebrate organisms that attach to surfaces individually or in colonies and filter-feed.

Biofouling: Also known as biological fouling. The unwanted accumulation of organisms on surfaces, where colonization can lead to negative consequences (e.g. on aquaculture cages, vessel hulls, pontoons, slipways).

Breakwaters: Also known as breakwalls, harbour walls and piers. Human-built structures placed in the marine environment to protect the shoreline from waves. Intertidal, subtidal or both.

Bryozoans: Members of the phylum Bryozoa. Mostly marine. Small invertebrate organisms that attach to surfaces in colonies, often forming hard crusts, and filter-feed.

Diatoms: Members of the phylum Ochrophyta. Single-celled microalgae found in floating phytoplankton and in 'biofilms' attached to surfaces.

Foundations: Also known as anchor/mooring blocks or weights. Human-built structures placed in the marine environment to provide secure seabed anchor points for infrastructure such as energy devices or navigation buoys. Subtidal.

Functional groups/richness: Species can be grouped according to their ecological role in the community instead of their taxonomy, e.g. their shape/structure, feeding strategy or position in the food chain. The functional richness of a community is defined as the amount of niche space occupied by the species within a community, but can informally refer to the number of functional groups present in a community.

Gabions: Also known as rock rolls, cages, baskets and reno mattresses. Human-built structures made from metal or mesh cages or baskets filled with rocks or other sediments, placed in the marine environment to retain slopes from erosion (but may have other uses). Intertidal or subtidal.

Groynes: Also known as groins, walls and fences. Human-built structures placed in the marine environment, perpendicular to the shoreline, to interrupt water flow and limit the movement of sediment. Intertidal, but sometimes extend into the subtidal.

Jetties: Also known as piers, docks and wharves. Human-built structures placed in the marine environment, connected to the shoreline, to provide access for sea-based activities. Over-water platforms supported by vertical poles or 'pilings' (see below) embedded in the seabed.

Mattresses: Articulated human-built structures laid over seabed infrastructure such as cables or pipelines to stabilize and protect them from damage. Subtidal, but sometimes extend into the intertidal.

Pilings: Also known as piles or poles. Vertical human-built structures placed in the marine environment and embedded in the seabed. Supporting columns for pontoons (see above), jetties (see above) and other platforms, or for wind turbines. Intertidal, subtidal or both.

Pontoons: Also known as floating docks, floating piers and floating jetties. Floating human-built structures placed in the marine environment, connected to the shoreline, to provide access for sea-based activities. Floatation maintains a fixed vertical relationship with the water surface regardless of the tidal level, making the submerged portions subtidal.

Scour protection: Human-built structures placed around the base of seabed infrastructure such as pilings or cables to reduce the removal of sediment by water flows that can destabilize structures. Subtidal, but sometimes extend into the intertidal.

Seawalls: Also known as sea walls, breakwalls and revetments. Human-built structures placed in the marine environment, parallel to the shoreline at the transition between land and sea, to retain and protect the land against wave action, flooding and/or coastal erosion. Intertidal, but sometimes extend into the subtidal.

Settlement plates: Also known as tiles, panels or experimental plots. Many marine ecological experiments use these to test hypotheses when it is not possible to replicate experiments at larger scales (e.g. multiple replicate seawalls with different treatments).

Sound (as used in this synopsis, i.e. Puget Sound): A large sea or ocean inlet containing large islands.

Strait: A narrow passage of water connecting two seas or other large bodies of water.

Appendix 2. List of Interventions for Enhancing the Biodiversity of Marine Artificial Structures

Intervention title	Definition and scale	Comments on definition	Section no. (No. studies)
Use environmentally-sensitive material on intertidal/subtidal artificial structures	'Environmentally-sensitive materials' are materials that seek to maximize environmental benefits and/or minimize environmental risks of marine engineering.	Defined by the authors.	Int 2.1 (8) Sub 3.1 (14)
Create textured surfaces (≤1 mm) on intertidal/subtidal artificial structures	'Texture' is micro-scale roughness applied to an entire surface that produces depressions and/or elevations ≤1 mm.	Defined by Strain *et al.* (2018).	Int 2.2 (4) Sub 3.2 (3)
Create natural rocky reef topography on intertidal/subtidal artificial structures	'Natural rocky reef topography' refers to the full fingerprint of substrate topography found in natural rocky habitats.	Defined by the authors.	Int 2.3 (2) Sub 3.3 (1)
Create pit habitats (1–50 mm) on intertidal/subtidal artificial structures	'Pit habitats' are depressions with a length to width ratio ≤3:1 and depth 1–50 mm.	Defined by Strain *et al.* (2018).	Int 2.4 (22) Sub 3.4 (1)
Create hole habitats (>50 mm) on intertidal/subtidal artificial structures	'Hole habitats' are depressions with a length to width ratio ≤3:1 and depth >50 mm that do not retain water during low tide.	Modified from "Subtidal holes" in Strain *et al.* (2018).	Int 2.5 (5) Sub 3.5 (3)
Create groove habitats (1–50 mm) on intertidal/subtidal artificial structures	'Groove habitats' are depressions with a length to width ratio >3:1 and depth 1–50 mm.	Modified from "Crevices" in Strain *et al.* (2018).	Int 2.6 (14) Sub 3.6 (2)
Create crevice habitats (>50 mm) on intertidal/subtidal artificial structures	'Crevice habitats' are depressions with a length to width ratio >3:1 and depth >50 mm.	Modified from "Crevices" in Strain *et al.* (2018).	Int 2.7 (0) Sub 3.7 (0)
Create 'rock pools' on intertidal artificial structures	'Rock pools' are depressions with a length to width ratio ≤3:1 and depth ≥50 mm that retain water during low tide.	Modified from "Intertidal water retaining features" in Strain *et al.* (2018).	Int 2.8 (18)
Create small adjoining cavities or 'swimthrough' habitats (≤100 mm) on intertidal/subtidal artificial structures	'Small adjoining cavities or 'swimthrough' habitats' are adjoining internal cavities sheltered from, but with access	Defined by the authors and advisory panel.	Int 2.9 (2) Sub 3.8 (4)

	to/from, outside the structure. Dimensions can vary but are ≤100 mm in any direction.		
Create large adjoining cavities or ‘swimthrough’ habitats (>100 mm) on intertidal/subtidal artificial structures	‘Large adjoining cavities or ‘swimthrough’ habitats’ are adjoining internal cavities sheltered from, but with access to/from, outside the structure. Dimensions can vary but are >100 mm in any direction.	Defined by the authors and advisory panel.	Int 2.10 (0) Sub 3.9 (2)
Create small protrusions (1–50 mm) on intertidal/subtidal artificial structures	‘Small protrusions’ are elevations with a length to width ratio ≤3:1 that protrude 1–50 mm from the substratum.	Modified from “Small elevations” in Strain *et al.* (2018).	Int 2.11 (2) Sub 3.10 (1)
Create large protrusions (>50 mm) on intertidal/subtidal artificial structures	‘Large protrusions’ are elevations with a length to width ratio ≤3:1 that protrude >50 mm from the substratum.	Modified from “Large elevations” in Strain *et al.* (2018).	Int 2.12 (2) Sub 3.11 (0)
Create small ridges or ledges (1–50 mm) on intertidal/subtidal artificial structures	‘Small ridges and ledges’ are elevations with a length to width ratio >3:1 that protrude 1–50 mm from the substratum.*	Modified from “Small elevations” in Strain *et al.* (2018).	Int 2.13 (4) Sub 3.12 (0)
Create large ridges or ledges (>50 mm) on intertidal/subtidal artificial structures	‘Large ridges and ledges’ are elevations with a length to width ratio >3:1 that protrude >50 mm from the substratum.*	Modified from “Large elevations” in Strain *et al.* (2018).	Int 2.14 (3) Sub 3.13 (0)
Create grooves *and* small protrusions, ridges or ledges (1–50 mm) on intertidal/subtidal artificial structures	‘Groove habitats’ are depressions with a length to width ratio >3:1 and depth 1–50 mm. ‘Small protrusions’ are elevations with a length to width ratio ≤3:1 that protrude 1–50 mm from the substratum. ‘Small ridges and ledges’ are elevations with a length to width ratio >3:1 that protrude 1–50 mm from the substratum.*	Combined intervention (see section 1.6.2d)	Int 2.15 (16) Sub 3.14 (3)
Create short flexible habitats (1–50 mm) on intertidal/subtidal artificial structures	‘Short flexible habitats’ are flexible protruding materials such as rope, ribbon or twine 1–50 mm in length.	Modified from “Soft structures” in Strain *et al.* (2018).	Int 2.16 (1) Sub 3.15 (3)
Create long flexible habitats (>50 mm) on intertidal/subtidal artificial structures	‘Long flexible habitats’ are flexible protruding materials such as rope, ribbon or twine >50 mm in length.	Modified from “Soft structures” in Strain *et al.* (2018).	Int 2.17 (1) Sub 3.16 (5)

Reduce the slope of intertidal/subtidal artificial structures.	'Reducing the slope' includes actions taken to reduce the inclination of structures without increasing the footprint, with the aim of enhancing their biodiversity.	Defined by the authors.	Int 2.18 (2) Sub 3.17 (0)
Transplant or seed organisms onto intertidal/subtidal artificial structures	'Transplanting or seeding species' includes actions taken to attach live organisms at any life stage onto structures, with the aim of generating self-sustaining populations.	Modified from "Habitat-forming taxa" in Strain *et al.* (2018).	Int 2.19 (10) Sub 3.18 (11)
Control or remove non-native or nuisance species on intertidal/subtidal artificial structures	'Controlling or removing non-native or nuisance species' includes actions taken to reduce the abundance of non-native or nuisance organisms on structures, with the aim of enhancing their biodiversity.	Defined by the authors.	Int 2.20 (0) Sub 3.19 (0)
Cease or alter maintenance activities on intertidal/subtidal artificial structures	'Ceasing or altering maintenance activities' includes actions taken to avoid or reduce the disturbance, damage or removal of native organisms from structures, with the aim of enhancing their biodiversity.	Defined by the authors.	Int 2.21 (0) Sub 3.20 (2)
Manage or restrict harvesting of species on intertidal/subtidal artificial structures	'Managing or restricting harvesting of species' includes actions taken to avoid or reduce the disturbance, damage or removal of native organisms from structures, with the aim of enhancing their biodiversity.	Defined by the authors.	Int 2.22 (2) Sub 3.21 (3)

'Int': intertidal; 'Sub': subtidal.

*On vertical surfaces, vertically-orientated elevations that fit these criteria are referred to as 'ridges', while horizontal ones are referred to as 'ledges'. On horizontal surfaces, these features are referred to as 'ridges' regardless of their orientation.

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Appendix 3. Literature found during searches for this synopsis

Table A3.1 (a) Number of publications found during literature searches, and (b) number included in the synopsis following screening.

Method	**(a) Number found and screened**	**(b) Number included**
(1) Keyword searches	831	54
(2) Review article searches*	110 additional to keyword searches (151 total)	14 additional to keyword searches (33 total)
(3) Conservation Evidence online database	2 additional to keyword searches (2 total)	2 additional to keyword searches (2 total)
(4) Advisory panel	37 additional to keyword searches (40 total)	16 additional to keyword searches (16 total)
Total	**980**	**86**

*We searched two recent eco-engineering reviews by Strain *et al.* (2018) and O'Shaughnessy *et al.* (2020).

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