Eco-engineered rock pools: a concrete solution to biodiversity loss and urban sprawl in the marine environment

LB Firth
KA Browne
AM Knights
SJ Hawkins
R Nash

Let us know how access to this document benefits you

General rights
All content in PEARL is protected by copyright law. Author manuscripts are made available in accordance with publisher policies. Please cite only the published version using the details provided on the item record or document. In the absence of an open licence (e.g. Creative Commons), permissions for further reuse of content should be sought from the publisher or author.

Take down policy
If you believe that this document breaches copyright please contact the library providing details, and we will remove access to the work immediately and investigate your claim.

Follow this and additional works at: https://pearl.plymouth.ac.uk/gees-research

Recommended Citation
Firth, L., Browne, K., Knights, A., Hawkins, S., & Nash, R. (2016) 'Eco-engineered rock pools: a concrete solution to biodiversity loss and urban sprawl in the marine environment', Environmental Research Letters, 11(9), pp. 94015-94015. IOP Publishing: Available at: 10.1088/1748-9326/11/9/094015
This Article is brought to you for free and open access by the Faculty of Science and Engineering at PEARL. It has been accepted for inclusion in School of Geography, Earth and Environmental Sciences by an authorized administrator of PEARL. For more information, please contact openresearch@plymouth.ac.uk.
Eco-engineered rock pools: a concrete solution to biodiversity loss and urban sprawl in the marine environment

Firth, LB

http://hdl.handle.net/10026.1/8398

10.1088/1748-9326/11/9/094015
Environmental Research Letters
IOP Publishing

All content in PEARL is protected by copyright law. Author manuscripts are made available in accordance with publisher policies. Please cite only the published version using the details provided on the item record or document. In the absence of an open licence (e.g. Creative Commons), permissions for further reuse of content should be sought from the publisher or author.
Eco-engineered rock pools: a concrete solution to biodiversity loss and urban sprawl in the marine environment

This content has been downloaded from IOPscience. Please scroll down to see the full text.

View the table of contents for this issue, or go to the journal homepage for more

Download details:

IP Address: 86.41.47.41
This content was downloaded on 14/09/2016 at 15:19

Please note that terms and conditions apply.
Eco-engineered rock pools: a concrete solution to biodiversity loss and urban sprawl in the marine environment

Louise B Firth\textsuperscript{1,2}, Keith A Browne\textsuperscript{2}, Antony M Knights\textsuperscript{3}, Stephen J Hawkins\textsuperscript{4,5} and Róisín Nash\textsuperscript{6}

\textsuperscript{1} School of Geography, Earth and Environmental Science, Plymouth University, Plymouth, UK
\textsuperscript{2} Ryan Institute, National University of Ireland Galway, Galway, Ireland
\textsuperscript{3} Marine Biology and Ecology Research Centre, School of Marine Science and Engineering, Plymouth University, Plymouth, UK
\textsuperscript{4} Marine Biological Association of the United Kingdom, Citadel Hill, Plymouth, UK
\textsuperscript{5} National Oceanography Centre Southampton, University of Southampton, Southampton, UK
\textsuperscript{6} Marine and Freshwater Research Center, Galway-Mayo Institute of Technology, Galway, Ireland

E-mail: louise.firth@plymouth.ac.uk

Keywords: alpha diversity, beta diversity, ecological engineering, habitat heterogeneity, ocean sprawl, reconciliation ecology, restoration ecology

Supplementary material for this article is available online

Abstract

In coastal habitats artificial structures typically support lower biodiversity and can support greater numbers of non-native and opportunistic species than natural rocky reefs. Eco-engineering experiments are typically trialed to succeed; but arguably as much is learnt from failure than from success. Our goal was to trial a generic, cost effective, eco-engineering technique that could be incorporated into rock armouring anywhere in the world. Artificial rock pools were created from manipulated concrete between boulders on the exposed and sheltered sides of a causeway. Experimental treatments were installed in locations where they were expected to fail and compared to controls installed in locations where they were expected to succeed. Control pools were created lower on the structure where they were immersed on every tidal cycle; experimental pools were created above mean high water spring tide which were only immersed on spring tides. We hypothesised that lower and exposed pools would support significantly higher taxon and functional diversity than upper and sheltered pools. The concrete pools survived the severe winter storms of 2013/14. After 12 months, non-destructive sampling revealed significantly higher mean taxon and functional richness in lower pools than upper pools on the exposed side only. After 24 months the sheltered pools had become inundated with sediments, thus failing to function as rock pools as intended. Destructive sampling on the exposed side revealed significantly higher mean functional richness in lower pools than upper pools. However, a surprisingly high number of taxa colonised the upper pools leading to no significant difference in mean taxon richness among shore heights. A high number of rare taxa in the lower pools led to total taxon richness being almost twice that of upper pools. These findings highlight that even when expected to fail concrete pools supported diverse assemblages, thus representing an affordable, replicable means of enhancing biodiversity on a variety of artificial structures.

1. Introduction

Urban sprawl—the expansion of humans away from central urban areas on land and its impacts on the environment has long been recognised as a global problem (Van Metre \textit{et al} 2000, Johnson 2001, Chadwick \textit{et al} 2006). Recently, ‘ocean sprawl’—the proliferation of artificial structures associated with coastal protection, shipping, aquaculture, and other coastal industries (coastal and marine infrastructure (CMI)) is gaining recognition as one of the biggest threats to marine ecosystems (Airoldi and Beck 2007, Firth \textit{et al} 2013a, Dafforn \textit{et al} 2015). Globally, coastal development is accelerating with many regions now dominated by artificial coastlines (e.g. Chapman 2003, Lai \textit{et al} 2015, Burt \textit{et al} 2013, Firth and Hawkins 2011,
Firth et al 2016, Knights et al 2016). Coasts are increasingly ‘hardened’ as new CMI is proliferating to support and protect burgeoning populations (Thompson et al 2002, Gerland et al 2014). There are however costs associated with replacing natural habitats with hard artificial structures including the loss of ecosystem services provided by those habitats (Beaumont et al 2007, Wyles et al 2014, Knights et al 2015), and alteration of connectivity patterns (Airoldi et al 2015, Bishop et al in press).

Sometimes CMI can provide important ecosystem services such as habitat or protection for species of commercial or conservation importance (e.g. Inger et al 2009, Toft et al 2013, Pearce et al 2014, Firth et al 2015, García-Gómez et al 2015). However, an increasing body of literature described how artificial habitats are poor surrogates for natural rocky reefs; supporting lower biodiversity (Moschella et al 2005, Firth et al 2013b), different communities (Chapman 2003, Evans et al 2016a), and promoting opportunistic and non-native species (Airoldi and Bulleri 2011, Firth et al 2011, Floerl et al 2009, Bracewell et al 2013).

Over 50% of CMI is constructed with Portland cement (Sharma 2009), which, despite being known to be toxic to some marine life (Nandakumar et al 2003, Togero 2006), can support diverse communities (e.g. Griffin et al 2010, Noël et al 2010). Colonising organisms with calcareous skeletons (e.g. oysters, corals) deposit calcium carbonate onto surfaces in a process termed biogenic build-up; which can contribute to the strength and durability of structures (Risinger 2012).

In response to mitigating the potential negative impacts of CMI, the field of eco-engineering has blossomed recently with a particular focus on demonstrating how CMI can be modified to enhance the habitat and promote biodiversity (reviewed in Dyson and Yocum 2015, Dafforn et al 2015, Firth et al 2016), in part, off-setting the negative consequences of lost natural habitat. Techniques include retrofitting novel microhabitats, such as rock pools, pits and surface roughness (Chapman and Blockley 2009, Martins et al 2010, Browne and Chapman 2014, Coombes et al 2015, Evans et al 2016a), deployment of precast concrete units (Perkol-Finkel and Sella 2014, 2015, Firth et al 2014a, Sella and Perkol-Finkel 2015, Loke and Todd 2016) and the transplantation of habitat-forming species onto artificial structures (Perkol-Finkel et al 2012, Ng et al 2015, Ferrario et al 2016).

The majority of eco-engineering experiments has been implemented at small spatial scales, with poor replication, limited monitoring and understandably, designed to succeed. Thanks to the recent surge of research in this area coastal managers are now faced with a toolkit of potential mitigation and reconciliation options when planning coastal infrastructure (see Dafforn et al 2015, Firth et al 2016 for reviews). However, all too often decisions about ’ticking the green box’ are made without consulting ecologists and thus projects can potentially fail, wasting taxpayer’s money. Perhaps even more serious is the possibility that the promise of incorporating eco-engineering into development plans may be used as a ploy for getting the green light on harmful coastal developments. Arguably, it is more important to know when schemes fail than when they succeed, as greater knowledge is gained from failure than from success.

To date, eco-engineering experiments conducted in the intertidal zone are largely carried out below mean high water spring tide (MHWS) where biodiversity enhancement is likely to be most successful due to more favourable abiotic conditions lower on the shore (Raffaelli and Hawkins 1996, Firth et al 2013b). Furthermore, trials are typically carried out under particular wave exposure conditions (e.g. Browne and Chapman 2011, Firth et al 2014a, Evans et al 2016a) making it difficult to generalise about success outside of the environmental conditions experienced at that location. The primary aim of our study was to trial a novel eco-engineering technique under conditions in which it was expected to succeed and compare the outcomes to treatments installed in conditions where it was expected to fail. The objective was two-fold. Firstly, to demonstrate proof-of-concept of a new technique, but secondly and perhaps more importantly, to identify where such an approach may not be suitable, thereby minimising the likelihood of investment in unsuitable technology and preserving consumer confidence (e.g. Knights et al 2014) in reconciliation and conservation efforts.

Artificial rock pools were created from concrete manipulated between boulders beneath the precast concrete hollow-core Shepherd Hill energy dissipation (SHED) units on a causeway in Galway Bay, Ireland (figure 1). The causeway offered a unique opportunity to trial the same technique under different environmental conditions simultaneously within a small spatial area. The west side of the causeway is exposed to swells coming in from the Atlantic Ocean, whilst the eastern side is relatively sheltered in comparison. Biodiversity patterns differ with varying levels of wave exposure (Ballantine 1961) with sheltered pools typically supporting lower biodiversity than moderately exposed rock pools (Firth, pers. obs.). The SHED units on the causeway were arranged in straight horizontal lines at different shore heights. Previous studies have found an inverse relationship between biodiversity and shore height in artificial habitats (Firth et al 2013b, Browne and Chapman 2014), but no studies to date have investigated the combined effects of wave exposure and shore height on colonisation of eco-engineered habitats.

The design of the causeway enabled the testing of the efficacy of the artificial rock pools among two different wave exposures (‘exposed’ and ‘sheltered’) and two different shore heights (‘lower’ and ‘upper’) simultaneously. Reference (control) pools were installed lower on the structure where they were immersed on every tidal cycle and experimental pools installed
above MHWS, which were only immersed on spring tides. Upper pools were deliberately placed above MHWS with an expectation of failure (i.e. very few species were expected to colonise the upper pools). We tested two hypotheses: (1) lower pools would support greater taxon and functional richness and different taxon and functional composition than upper pools; (2) exposed pools would support greater taxon and functional richness and different taxon and functional composition than sheltered pools.

2. Study area

Galway Bay is a large marine-dominated bay on the west coast of Ireland. The total length of coastline in Galway Bay from Cloghmore, Co. Galway (53°13’N, 9°31’W) to Black Head, Co. Clare (53°09’N, 9°15’W) is 191 km, comprising ~180 km (93%) natural and ~12.6 km (7%) artificial coastline. The present length of coastline within the Galway City limits is ~23 km, comprising ~13 km (55%) natural and ~10.5 km (45%) artificial coastline. There are plans to expand the existing harbour to create commercial quays and a deep water docking facility (figure 2); the proposed development will increase the extent of the artificial coastline from 10.5 to 14.6 km (53%), an increase of 39%.

2.2. Experimental set up

2.2.1. Causeway engineering details

Mutton Island is connected to Galway City by a 0.8 km causeway (figure 2, 53°15’27”N, 09°03’18” W). The top of the causeway is 1.4 m above MHWS. Precast concrete toe units hold concrete SHED units in place, which are fronted with rock armour toes (figure 1). In June 2013, artificial rock pools were created by pouring quick-drying concrete around buckets in the base of the SHED units (figure 1(c)). The concrete was sufficiently coarse and dry enough not to run among the boulders beneath, but could be easily manipulated to fill the interstitial spaces among boulders. Once the concrete had set, the buckets were removed to reveal water-retaining depressions (hereafter referred to as a ‘pool’) of opening diameter 13–14 cm, bottom diameter 10.6 cm and 10–12 cm depth (~1250 cm³ volume). In total, 80 pools were created: 20 upper (0.4 m above MHWS) and 20 lower (1.9 m below MHWS), and replicated on both the eastern (hereafter ‘sheltered’) and western (hereafter ‘exposed’) sides of the causeway.

2.2.2. Sampling

After 12 months (May 2014), all colonising organisms were sampled visually in situ using non-destructive techniques. After 24 months (May 2015), all colonising organisms were removed from the pools and identified to the lowest possible taxonomic level in the
laboratory. Flora and fauna were grouped into functional groups based on morphological features (Eriksen et al. 2002) and feeding strategies respectively.

2.3. Statistical analyses
To address hypothesis 1—that lower pools would support greater taxon and functional richness and different taxon and functional composition than upper pools, data were treated separately for the 12 month and 24 month sampling periods. To address hypothesis 2—that exposed pools would support greater taxon and functional richness and different taxon and functional composition than pools on the sheltered side, only the data collected after 12 months was used as pools on the sheltered side became inundated with sediment some time between 12 and 24 months and were therefore omitted from analyses.

Analysis of variance (ANOVA) and permutational analysis of variance (PERMANOVA, Anderson 2001) were used to test hypotheses 1 and 2. ANOVA was used to test for differences in mean taxon and functional richness based on presence/absence data. PERMANOVA was used to test for differences in multivariate taxon and functional composition, based on 9999 unrestricted permutations of raw presence/absence data. Percentage contributions of individual taxa and functional groups to dissimilarity between communities were calculated using SIMPER (Clarke 1993). SIMPER analysis in the PRIMER package was used to assess which species were most influential in causing similarity among plots within treatments and dissimilarity among different treatments (Clarke and Warwick 1994). In all cases, the same design was used for ANOVA and PERMANOVA, but with separate designs for 12 and 24 month comparisons. For 12 months, a two-way crossed design was used, with fixed factors Exposure and Shore Height. The 24 month data were analysed using a one-way design with fixed factor Shore Height.

Using 24 month data, the likelihood of species occurring in pools at different shore heights (probability of occurrence) was estimated from incidence matrices (i.e. data describing the presence or absence of a species in a pool, (Dorazio et al. 2011)). The sum presence of a species over the number of pools sampled is used to calculate the likelihood of occurrence of a species. This uses binary presence–absence data and not abundance data. Random resampling with replacement was used to generate species accumulation curves to estimate the predicted number of taxa with an increasing number of pools.

Beta diversity which has been defined as the ratio between gamma (regional) and alpha (local) diversities (Whittaker 1960) can also be used as a measure of community heterogeneity (Tuomisto 2010a, 2010b). Here, we used the Sørensen index of dissimilarity (Whittaker 1960) to compare communities in terms of the number of shared and unique taxa recorded in different pools within shore heights. Larger values of beta diversity (i.e. those that tend toward 1) indicate greater dissimilarity between pools. Species occurrence probabilities, species accumulation curves and Sørensen
index estimates were calculated using the R package ‘Vegan’ (Oksanen et al. 2016). Pairwise comparisons of beta diversity between pool pairs after 24 months are shown using correlation plots, and non-metric multi-dimensional scaling used to show dissimilarity in taxon and functional compositions between exposure and shore heights after 12 and 24 month periods respectively.

3. Results

3.1. Biotic colonisation of concrete pools after 12 months

Non-destructive sampling identified 19 taxa from 8 functional groups (table 1(a)). All 19 taxa were recorded in exposed pools in comparison to only 4 in sheltered pools. A greater number of taxa were found in the lower (15) than upper pools (10) on the exposed side.

3.1.1. Taxon and functional richness

There was a significant interaction between exposure and shore height for both taxon and functional richness (table 2(a), figure 3(a)). On the exposed side lower pools supported significantly greater mean taxon (4.6) and functional richness (3.6) than upper pools (3.7 and 2.3 respectively), whilst on the sheltered side there was no significant difference between lower and upper pools (figure 3(a)) in either taxon or functional richness.

3.1.2. Taxon and functional composition

There was a significant interaction between exposure and shore height for both taxon and functional compositions (table 2(b)). Post hoc pairwise tests revealed significant differences in taxon and community composition (table 2(b), figures 4(a) and (b)) between exposures for both lower and upper pools. Similarly, pairwise tests revealed significant differences in taxon and functional composition between shore heights for exposed but not for sheltered pools (figures 4(a) and (b)). It is clear that there was little variance in the taxon and functional composition in sheltered pools (figures 4(a) and (b)), but far more variance in exposed pools (figures 4(a) and (b)). As predicted, SIMPER analysis revealed that there were greater numbers of taxa and functional groups associated with lower than upper pools and exposed than sheltered pools (table S1). The most diverse habitat was lower exposed pools.

3.2. Biotic colonisation of concrete pools after 24 months

Following the 12 month sampling, the sheltered pools became inundated with sediments. Consequently, we refer only to the exposed pools here. After 24 months, destructive sampling yielded 72 taxa across 11 functional groups (table 1(b)). 63 taxa were found in lower and 37 in upper pools (figure 3(c)), two of which classified as non-native: the algae Colpomenia peregrina and the barnacle, Austrominius modestus. Of the 72 taxa recorded, 35 (48.6%) and 9 (12.5%) were unique to the lower and upper pools respectively (table 1, figures 4(d) and (e)).

3.2.1. Taxon and functional richness

There was no significant difference in mean taxon richness between lower (16.9) and upper pools (13.9) (table 2(c), figure 3(b)); but lower pools did support

### Table 1. Summary of the number of taxa in each functional group (a) after 12 months and (b) after 24 months in lower and upper pools on exposed and sheltered sides of the Mutton Island causeway, Galway Bay, Ireland.

| Functional group | Causeway | Exposed only | Sheltered only | Exposed | Total | Lower | Upper |
|------------------|----------|--------------|----------------|---------|-------|-------|-------|
|                  | Total    | Total        | Lower          | Total   | Lower | Upper |
| Thin filamentous | 3        | 3            | 2              | 3       | 1     | 1     |
| Corticated foliose | 1        | 1            | 1              | 1       | 0     | 0     |
| Corticated macrophyte | 2        | 2            | 2              | 0       | 0     | 0     |
| Leathery macroalga  | 0        | 0            | 0              | 0       | 0     | 0     |
| Calcareous algae   | 1        | 1            | 1              | 0       | 0     | 0     |
| Suspension feeder   | 6        | 6            | 5              | 2       | 0     | 0     |
| Grazer             | 3        | 3            | 3              | 1       | 0     | 1     |
| Detritivore        | 1        | 1            | 1              | 1       | 0     | 1     |
| Carnivore          | 0        | 0            | 0              | 0       | 0     | 0     |
| Omnivore           | 2        | 2            | 2              | 0       | 1     | 1     |
| Other              | 0        | 0            | 0              | 0       | 0     | 0     |
| Total taxon richness | 19       | 19           | 15             | 10      | 4     | 2     |
| Total functional richness | 8         | 8            | 8              | 5       | 4     | 2     |
| Unique taxon richness | 35       | 35           | 35             | 35      | 11    | 11    |
| Rare taxon richness | 13        | 13           | 13             | 13      | 4     | 4     |
significantly greater mean functional richness than upper pools (8.5 versus 6.7, table 2(c), figure 3(b)).

The community composition in lower pools was more variable than in upper pools (figures 4(c) and (d)), although this variation is largely attributed to a large number of taxa with relatively low probabilities of occurrence (table 3, figure 5; species ranks 38–63). The species with a higher probability of occurrence (>55%) tended to occur in both lower and upper pools (with the exception of *Rhizoclonium riparium* and *Ulva linza*). For species less likely to occur (<55% probability of occurrence), 30% were unique to upper and 67% were unique to lower pools.

Unsurprisingly, mean beta diversity of the upper pools (0.39) was lower than that of the lower pools (0.54). Pairwise comparisons of beta diversity between individual pools revealed considerable differences in community composition, even between lower pools established in the same SHED unit (figure 6; L8A versus L8B—β = 0.69), whereas upper pools were more similar (figure 6).

Comparing taxon accumulation across pools, the number of taxa in upper pools approached asymptote with the inclusion of 11 pools (32 taxa), with only an additional five taxa accumulating across the remaining nine pools (20 pools: 37 taxa). Conversely, for lower

| Source                  | Taxon |        |        |        | Functional |        |        |        |
|-------------------------|-------|--------|--------|--------|------------|--------|--------|--------|
|                         | df    | MS     | F      | P      |            | MS     | F      | P      |
| Exposure                | 1     | 33.5868| 415.9  | 0      | 63.0125    | 81.65  | 0      |        |
| Shore height            | 1     | 0.0937 | 1.16   | 0.2848 | 5.5125     | 7.14   | 0.0092 |        |
| Exposure X shore height | 1     | 0.3796 | 4.7    | 0.0033 | 7.8125     | 10.12  | 0.0021 |        |
| Residual                | 76    | 0.0808 |        |        | 0.7717     |        |        |        |
| Total                   | 79    |        |        |        |            |        |        |        |
| Transformation          |       | Ln(X)  |        |        | None       |        |        |        |
| Cochran’s C             |       | NS     |        |        | P < 0.01   |        |        |        |

| Source                  | Taxon |        |        |        | Functional |        |        |        |
|-------------------------|-------|--------|--------|--------|------------|--------|--------|--------|
|                         | df    | MS     | F      | P      |            | MS     | F      | P      |
| Exposure                | 1     | 49779  | 49.661 | <0.0001| 44.405     | 79.35  | <0.0001|        |
| Shore height            | 1     | 11624  | 11.597 | <0.0001| 6769.6     | 12.097 | <0.0001|        |
| Exposure X shore height | 1     | 16783  | 16.744 | <0.0001| 8443.6     | 15.089 | <0.0001|        |
| Residual                | 76    | 1002.4 |        |        | 559.61     |        |        |        |
| Total                   | 79    |        |        |        |            |        |        |        |
| Transformation          |       | SQRT   |        |        | SQRT       |        |        |        |

| Source                  | Taxon |        |        |        | Functional |        |        |        |
|-------------------------|-------|--------|--------|--------|------------|--------|--------|--------|
|                         | df    | MS     | F      | P      |            | MS     | F      | P      |
| Shore height            | 1     | 90     | 4.55   | 0.0394| 18.225     | 8.33   | 0.0064 |        |
| Residual                | 38    | 19.7789|        |        | 2.1882    |        |        |        |
| Total                   | 39    |        |        |        |            |        |        |        |
| Transformation          |       | None   |        |        | None       |        |        |        |
| Cochran’s C             |       | P < 0.01|        |        | NS         |        |        |        |

| Source                  | Taxon |        |        |        | Functional |        |        |        |
|-------------------------|-------|--------|--------|--------|------------|--------|--------|--------|
|                         | df    | MS     | F      | P      |            | MS     | F      | P      |
| Shore height            | 1     | 18551  | 15.6   | <0.0001| 3534.2     | 10.593 | <0.0001|        |
| Residual                | 38    | 1189.2 |        |        | 333.64     |        |        |        |
| Total                   | 39    |        |        |        |            |        |        |        |
| Transformation          |       | SQRT   |        |        | SQRT       |        |        |        |

Table 2. (a) ANOVA comparing taxon and functional richness and (b) PERMANOVA comparing taxonomic and functional composition among exposures and shore heights after 12 months. (c) ANOVA of taxon and functional richness; (d) PERMANOVA comparing taxonomic and functional composition among shore heights after 24 months. Where Cochran’s C was significant α was set to 0.01 (Underwood 1997). Significant P-values are in bold.
pools, the number of species did not reach asymptote (20 pools: 63 taxa; figure 3(c)) with new taxa recorded in nearly every additional pool.

3.2.2. Taxon and functional composition
Composition differed among shore heights for both taxon and functional groups (figures 4(c) and (d)). Of the top ten taxa contributing most to the dissimilarities, five were positively associated with lower and five were positively associated with upper pools (table S3). Of the functional groups that contributed most to the dissimilarities, all but one (corticated foliose) were positively associated with lower pools.

4. Discussion
In this experiment we installed novel concrete pools at two shore heights (lower and upper) on two sides (exposed and sheltered) of an intertidal causeway. We hypothesised that lower and exposed pools would support greater diversity (i.e. mean taxon and functional richness) than upper and sheltered pools. In short we expected the upper and sheltered pools to 'fail'.

If the experiment was terminated after 12 months (and used non-destructive sampling techniques), we would have concluded that a total of 19 species...
Table 3. Estimated probabilities of occurrence (with 95% confidence intervals) for intertidal marine species occurring in low shore and high pools. Probabilities are estimated at the average value. Shading indicates species not recorded at shore height.

| Species                          | Lower site occurrence probability | Upper site occurrence probability |
|----------------------------------|-----------------------------------|----------------------------------|
|                                  | Median   | 2.5% | 97.5% | Rank | Median   | 2.5% | 97.5% | Rank |
| Actinia equina                   | 0.124    | 0.118 | 0.194 | 31   | 0.000    | 0.000 | 0.000 | —    |
| Actinia fragacea                 | 0.049    | 0.045 | 0.065 | 54   | 0.000    | 0.000 | 0.000 | —    |
| Amathia sp.                      | 0.449    | 0.422 | 0.513 | 17   | 0.000    | 0.000 | 0.000 | —    |
| Amphipod indet                   | 0.197    | 0.190 | 0.248 | 27   | 0.099    | 0.097 | 0.138 | 26   |
| Anurida maritima                 | 0.773    | 0.673 | 0.778 | 9    | 0.646    | 0.610 | 0.666 | 11   |
| Apherusa jurinee                 | 0.039    | 0.028 | 0.071 | 60   | 0.238    | 0.231 | 0.304 | 19   |
| Aquatic larvae (Chironomidae)    | 0.017    | 0.009 | 0.058 | 63   | 0.058    | 0.054 | 0.088 | 32   |
| Austrominia modesta              | 0.670    | 0.589 | 0.682 | 12   | 0.040    | 0.035 | 0.061 | 36   |
| Bryopsis sp.                     | 0.220    | 0.217 | 0.285 | 24   | 0.000    | 0.000 | 0.000 | —    |
| Bryozoon indet                   | 0.086    | 0.074 | 0.146 | 41   | 0.125    | 0.119 | 0.165 | 23   |
| Calliostoma occidentale          | 0.063    | 0.054 | 0.107 | 47   | 0.000    | 0.000 | 0.000 | —    |
| Callithamnion corymbosum         | 0.780    | 0.701 | 0.782 | 8    | 0.232    | 0.228 | 0.276 | 20   |
| Carcinus maenas                  | 0.706    | 0.616 | 0.708 | 11   | 0.589    | 0.554 | 0.601 | 14   |
| Ceramium sp.                     | 0.501    | 0.460 | 0.584 | 16   | 0.000    | 0.000 | 0.000 | —    |
| Ceramium strictum                | 0.000    | 0.000 | 0.000 | —    | 0.115    | 0.109 | 0.157 | 25   |
| Chaetomorpha linium              | 0.823    | 0.692 | 0.813 | 6    | 0.713    | 0.674 | 0.723 | 8    |
| Chaetomorpha rupestris           | 0.000    | 0.000 | 0.000 | —    | 0.084    | 0.078 | 0.116 | 28   |
| Chondria daeophysilla            | 0.093    | 0.087 | 0.143 | 39   | 0.000    | 0.000 | 0.000 | —    |
| Chondrus crispus                 | 0.932    | 0.875 | 0.940 | 2    | 0.045    | 0.036 | 0.065 | 34   |
| Chordaria flagelliformis         | 0.000    | 0.000 | 0.000 | —    | 0.254    | 0.242 | 0.311 | 18   |
| Cirriformia tentaculata          | 0.114    | 0.099 | 0.160 | 33   | 0.000    | 0.000 | 0.000 | —    |
| Cladophora raphioptera           | 0.833    | 0.753 | 0.844 | 4    | 0.640    | 0.590 | 0.662 | 13   |
| Cladophora sp.                   | 0.074    | 0.064 | 0.121 | 44   | 0.084    | 0.074 | 0.133 | 29   |
| Colpomenia peregrina             | 0.000    | 0.000 | 0.000 | —    | 0.086    | 0.081 | 0.115 | 27   |
| Corallina officinalis            | 0.288    | 0.278 | 0.344 | 20   | 0.000    | 0.000 | 0.000 | —    |
| Coryne pusilla                   | 0.243    | 0.236 | 0.317 | 23   | 0.000    | 0.000 | 0.000 | —    |
| Echinogammarus stoerensis        | 0.092    | 0.082 | 0.134 | 40   | 0.000    | 0.000 | 0.000 | —    |
| Ectocarpus sp.                   | 0.941    | 0.905 | 0.943 | 1    | 0.874    | 0.838 | 0.881 | 5    |
| Electra pilosa                   | 0.039    | 0.028 | 0.071 | 60   | 0.000    | 0.000 | 0.000 | —    |
| Eusyllis bomastrandi             | 0.120    | 0.113 | 0.158 | 32   | 0.000    | 0.000 | 0.000 | —    |
| Fabricia stellaris               | 0.162    | 0.152 | 0.210 | 29   | 0.000    | 0.000 | 0.000 | —    |
| Flustrellidra hispida            | 0.086    | 0.074 | 0.146 | 41   | 0.000    | 0.000 | 0.000 | —    |
| Fucus serratus                   | 0.095    | 0.087 | 0.137 | 38   | 0.000    | 0.000 | 0.000 | —    |
Table 3. (Continued.)

| Species                          | Median | 2.5% | 97.5% | Rank | Median | 2.5% | 97.5% | Rank |
|----------------------------------|--------|------|-------|------|--------|------|-------|------|
| Fucus sp.                        | 0.133  | 0.128| 0.222 | 30   | 0.962  | 0.947| 0.965 | 1    |
| Fucus vesiculosus                | 0.055  | 0.044| 0.089 | 51   | 0.000  | 0.000| 0.000 | —    |
| Gibbula umbilicalis              | 0.060  | 0.052| 0.086 | 48   | 0.000  | 0.000| 0.000 | —    |
| Harnia imbricata                 | 0.162  | 0.149| 0.218 | 28   | 0.000  | 0.000| 0.000 | —    |
| Jaera albifrons                  | 0.057  | 0.051| 0.114 | 50   | 0.000  | 0.000| 0.000 | —    |
| Laminaria sp. jex                | 0.870  | 0.816| 0.880 | 3    | 0.201  | 0.196| 0.257 | 21   |
| Laomedea flexuosa                | 0.274  | 0.260| 0.347 | 21   | 0.000  | 0.000| 0.000 | —    |
| Lepidochitona cinerea            | 0.000  | 0.000| 0.000 | —    | 0.022  | 0.015| 0.056 | 37   |
| Lipophrys pholis                 | 0.024  | 0.014| 0.048 | 62   | 0.000  | 0.000| 0.000 | —    |
| Littorina littorea               | 0.660  | 0.558| 0.673 | 13   | 0.490  | 0.466| 0.526 | 15   |
| Myriogramme sp.                 | 0.049  | 0.040| 0.081 | 55   | 0.000  | 0.000| 0.000 | —    |
| Mytilus edulis                   | 0.054  | 0.045| 0.074 | 53   | 0.000  | 0.000| 0.000 | —    |
| Mytilus edulis                  | 0.718  | 0.640| 0.737 | 10   | 0.096  | 0.075| 0.090 | 3    |
| Nucella lapillus egg             | 0.107  | 0.090| 0.142 | 35   | 0.000  | 0.000| 0.000 | —    |
| Palamaria palmata                | 0.304  | 0.304| 0.391 | 19   | 0.000  | 0.000| 0.000 | —    |
| Plagiodon tubulosa               | 0.042  | 0.035| 0.073 | 59   | 0.122  | 0.119| 0.171 | 24   |
| Polydora fallica                 | 0.049  | 0.040| 0.081 | 55   | 0.000  | 0.000| 0.000 | —    |
| Perina alvina                    | 0.055  | 0.044| 0.089 | 51   | 0.000  | 0.000| 0.000 | —    |
| Plagiodon tubulosa               | 0.396  | 0.366| 0.463 | 18   | 0.043  | 0.037| 0.063 | 35   |
| Polydora fallica                 | 0.065  | 0.059| 0.086 | 45   | 0.000  | 0.000| 0.000 | —    |
| Polyplithon sp.                  | 0.823  | 0.733| 0.826 | 5    | 0.690  | 0.621| 0.696 | 10   |
| Polysiphonia sp. B               | 0.113  | 0.096| 0.140 | 34   | 0.058  | 0.054| 0.088 | 32   |
| Procerodes littoralis            | 0.083  | 0.080| 0.128 | 43   | 0.000  | 0.000| 0.000 | —    |
| Red filamentosa                  | 0.000  | 0.000| 0.000 | —    | 0.291  | 0.283| 0.337 | 17   |
| Rhizoclonium riparium            | 0.000  | 0.000| 0.000 | —    | 0.888  | 0.854| 0.896 | 4    |
| Rhodothamnium floridula          | 0.046  | 0.041| 0.068 | 57   | 0.000  | 0.000| 0.000 | —    |
| Rissoella sp.                    | 0.064  | 0.058| 0.120 | 46   | 0.000  | 0.000| 0.000 | —    |
| Securifera fuciformes            | 0.203  | 0.195| 0.267 | 26   | 0.000  | 0.000| 0.000 | —    |
| Spirobranchus triquetre          | 0.571  | 0.507| 0.617 | 15   | 0.340  | 0.334| 0.399 | 16   |
| Spiorbis sp.                     | 0.821  | 0.740| 0.824 | 7    | 0.644  | 0.598| 0.652 | 12   |
Table 3. (Continued.)

| Species             | Lower site occurrence probability | Upper site occurrence probability |
|---------------------|----------------------------------|----------------------------------|
|                     | Median  | 2.5%  | 97.5% | Rank | Median  | 2.5%  | 97.5% | Rank |
| *Talitrus saltator* | 0.044   | 0.034 | 0.108 | 58   | 0.000   | 0.000 | 0.000 | —    |
| *Testudinalia testudinalis* | 0.000 | 0.000 | 0.000 | —    | 0.060   | 0.054 | 0.081 | 31   |
| *Ulva intestinalis*  | 0.105   | 0.097 | 0.154 | 36   | 0.856   | 0.802 | 0.860 | 6    |
| *Ulva lactuca*       | 0.606   | 0.525 | 0.624 | 14   | 0.915   | 0.897 | 0.922 | 2    |
| *Ulva linza*         | 0.000   | 0.000 | 0.000 | —    | 0.837   | 0.805 | 0.848 | 7    |
colonised the pools, both taxon and functional richness were significantly greater in lower and exposed pools than upper and sheltered pools (as hypothesised) and that composition differed among habitats (as hypothesised). These results would have provided additional evidence to support the assertion that these patterns occur in both natural (Hawkins and Hartnoll 1980, Raffaelli and Hawkins 1996) and artificial habitats (Browne and Chapman 2011, Firth et al 2013b).

By running the experiment for 24 months (and using destructive sampling), a very different picture emerged. The sheltered pools became inundated with sediment, thus failing to function as rock pools, instead supporting muddy habitats—which may be a desirable habitat to create in some locations. 72 taxa were observed on the exposed side alone; almost four times that observed across both exposures after 12 months. A surprisingly high number of taxa colonised the upper pools (37), including some species that are typically found in the lower intertidal and shallow subtidal (e.g. the kelp, Laminaria digitata). This diversity in the upper pools led to no significant difference in mean taxon richness among shore heights, despite the total number of taxa in lower pools (63) being almost double the upper pools (37); a pattern also observed in natural rock pools (Firth et al 2014b). The opposite pattern was observed for functional groups with little difference in the total number observed among shore heights (11 lower, 10 upper), but significantly higher mean functional richness in lower (8.1) than upper pools (6.7). Faunal groups (suspension feeders, grazers and carnivores) were particularly lacking in upper pools, whilst filamentous and foliose algal groups were better represented in upper compared to lower pools.

The differential results obtained among the 12 and 24 month surveys highlights the importance of sustained monitoring (Hawkins et al 2013a, 2013b, Mieszkowska et al 2014) and the most appropriate sampling technique and response variables for the question being asked. Species richness, which is indicative of alpha diversity, is a popular response variable considered in comparative ecological studies. However, in the context of habitat creation, the numbers of additional taxa and total richness revealed by taxon accumulation curves (indicative of beta diversity, Hawkins and Hartnoll 1980), might in fact be more meaningful than mean richness, which is often used in similar studies (e.g. Firth et al 2014a). Furthermore, the taxon accumulation curves revealed that the number of taxa in upper pools approached asymptote at 11 pools whilst the number of taxa kept increasing for lower pools. In this context, if the management goal was to enhance biodiversity compared to the surrounding structure, then fewer than 20 pools was likely to achieve this (particularly higher up), thus reducing the cost of the intervention (see section below on cost). If enhancement of rare taxa was the management goal, then the greater the number of pools, the greater the likelihood of colonisation by rare taxa.

Figure 4. MDS plots of taxon (a), (c) and functional composition (b), (d) compared among the lower (black shapes) and upper pools (grey shapes) on exposed (squares) and sheltered (triangles) sides after 12 months (a), (b); non-destructive sampling, (n = 20) and 24 months (c), (d); destructive sampling, (n = 20).
The cessation of experiments at different time points can yield variable results particularly when considering groups that exhibit seasonal fluctuations (e.g. O’Connor and Crowe 2005, Mrowicki et al. 2015). The importance of judging success of rehabilitation studies over extended time periods has been repeatedly voiced (Lirman and Miller 2003, Cunha et al. 2012), but often monitoring programmes are finite, and timescales are determined by funding cycles. For example, a seagrass restoration project in Florida was reported as...
unsuccessful 3 years post-planting in 2005 (Bell et al. 2008). Sustained monitoring of the plots for a further 4 years meant that the researchers captured a rapid increase in seagrass growth and cover, thus leading them to reverse the earlier evaluation of project ‘success’ (Bell et al. 2014). We advise that natural seasonal and successional cycles are considered and that monitoring programmes be tailored appropriately for the system which has undergone rehabilitation.

The consideration of taxon accumulation across the number of pools revealed that greater numbers of rare species were found in lower rather than upper pools. In a study of intertidal microgametopods on ‘artificial units of habitat’, Chapman and Underwood (2008) found that rare species showed greater small-scale (20 cm) and less large-scale (4 km) variation than common species. If persistence of rare species in artificial habitats can promote biotic resistance (Stachowicz et al. 2002, Arenas et al. 2006) or resilience to external stress (Benedetti-Cecchi et al. 2008), they may play a very important role in ecosystem functioning (Matias et al. 2012, Mrowicki et al. in press).

The colonisation of artificial structures by non-native species is one of the drivers of global biotic homogenisation (Mineur et al. 2012, Simkanin et al. 2013) and is one of the biggest concerns of coastal managers (Evans et al. 2016b). Traditional methods of mechanical and chemical removal can be expensive and have negative effects on the surrounding environment (Caffrey et al. 2010, Atalah et al. 2013). Recently there has been a move towards biological control through the introduction of natural predators (e.g. Atalah et al. 2014, 2015); or through the development of diverse and resilient communities—sensu the diversity resistance hypothesis (Elton 1958, Stachowicz et al. 2002, Arenas et al. 2006). Rare taxa are typically missing from artificial structures (Chapman 2003, Pister 2009). If eco-engineering can create novel habitats and encourage the settlement of rare taxa (like in the present study), the resistance of the engineered structures to biotic invasion may be enhanced (Stachowicz et al. 2002, Arenas et al. 2006).

Eco-engineering can also be used to increase abundance of target species. Rock pools are known to be nursery habitats for limpets (Bowman and Lewis 1977) which are key grazers on natural rocky shores (Hawkins and Hartnoll 1983, O’Connor and Crowe 2005, Moore et al. 2007). Encouraging limpets can increase grazing pressure leading to removal of algae (Hawkins et al. 1983, Jonsson et al. 2006) which can pose a slip hazard to members of the public that inevitably climb on the structures. Long-term studies of succession on natural and artificial rocky shores in Europe show that early colonisation is dominated by ephemeral algae, then fucoids with later colonising limpets grazing down the algae (Southward and Southward 1978, Hawkins et al. 1983). Engineering pits and crevices or nursery pools (Martins et al. 2010, Chapman and Underwood 2011, Skov et al. 2011, Firth et al. 2014a) can all increase grazing pressure, which can break inhibition during succession by ephemeral algae (Sousa 1979, Hawkins 1981).

Eco-engineering is expensive and the price is typically borne by the taxpayer. Managers, therefore, seek assurance that any enhancements will be resilient to tidal and storm damage, or more simply function in a manner that is deemed beneficial. Lack of confidence in the approach could undermine future rehabilitation efforts (Knights et al. 2014). During winter 2013/14, Western Europe experienced an unprecedented sequence of stormy conditions (Huntingford et al. 2014, Matthews et al. 2014); the west coast of Ireland was particularly badly affected. The artificial rock pools in Galway Bay were unaffected (100% survival), suggesting that this eco-engineering option may represent a long-term solution that will be resilient to storm damage.

This experiment was conducted at a single site, but comprised the creation of 80 concrete pools; 20 replicates of each combination of exposure and shore height; making it a comparably well-replicated study. Due to the public nature of the site, it was a requirement to get contractors to set up the experiment, making it more expensive than if we had done it ourselves. This involved the hiring of a digger, truck, cement mixer and three contractors (labour) for 5 days in addition to the cost of the concrete mixing materials. The cost was relatively low (∼€38 per pool) in comparison to other methods, such as the use of drill-coring (Evans et al. 2016a, ∼€60 per pool); although both methods are relatively low-cost and long-term options.

In the Galway Bay experiment, the concrete was manipulated among the SHED units and rock armouring. This method could easily be adapted by simply pouring concrete among the boulders of any rock armouring structure. We advocate that the manipulated units should be interspersed across a structure, rather than concentrated in a local area, as this may have implications for the porosity and thus the wave dissipation function of the structure. The manipulation of concrete to create pools and other microhabitats represents an affordable, robust, universally applicable method that could be incorporated into rock armouring anywhere in the world.

Acknowledgments

The authors wish to thank Martina Mulligan (AECOM), Billy Dunne, Stephen Walsh and Helen McGuinness (Galway City Council) for facilitating this work and David Mauerhofer, Gemma Gale, Lisa Grant and Amy Spain-Butler for helping out with fieldwork and initial data collection. A special thank you to Bernard, Peter and Paddy from Kevin Madden Ltd for constructing the pools. Keith Browne was supported by a National University of Ireland Galway, College of Science Summer Internship. The authors would also like to thank Tim Absalom (Plymouth University) for...
creating the map of Galway Bay and to Nessa O’Connor and Jonathan Ellis for commenting on earlier drafts of this manuscript.

References
Airoldi L and Beck M W 2007 Loss, status and trends for coastal marine habitats of Europe Oceanogr. Mar. Biol. Annu. Rev. 45 345–405
Airoldi L and 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 and Rius M 2015 Corridors for aliens but not for natives: effects of marine urban sprawl at a regional scale Divers. Distrib. 21 755–68
Anderson M J 2001 A new method for non-parametric multivariate analysis of variance Austr. Ecol. 26 32–46
Atalah J, Hopkins G A, Fletcher L M, Castinel A and Forrest B M 2013 Evaluation of the sea anemone Anthothoe albocincta as an augmentative biocontrol agent for biofouling on artificial structures Biofouling 29 359–71
Atalah J, Hopkins G A, Fletcher L M, Castinel A and Forrest B M 2015 Concepts for biocontrol in marine environments: is there a way forward? manage Biol. Invas. 6 1–12
Atalah J, Newcombe E M, Hopkins G A and Forrest B M 2014 Potential biocontrol agents for biofouling on artificial structures Biofouling 30 999–1010
Ballantine W J 1961 A biologically-defined exposure scale for the comparative description of rocky shores Field Stud. 1 73–84
Benedetti-Cecchi L, Bertocci I, Vaselli S, Maggi E and Bulleri F 2008 Identiﬁcation and deﬁnition of goods and services provided by marine biodiversity: implications for the ecosystem approach Mar. Poll. Bull. 54 253–65
Bell S S, Middlebrooks M L and Hall M O 2014 The value of long-term assessment of restoration: support from a seagrass investigation Restor. Ecol. 2 304–10
Bell S S, Tewﬁk A, Hall M O and Fonseca M S 2008 Evaluation of seagrass planting and monitoring techniques: implications for assessing recovery times and habitat equivalency analysis Restor. Ecol. 16 407–16
Benedetti-Cecchi L, Bertocci I, Vasselli S, Maggi E and Bulleri F 2008 Neutralism and the response of rare species to environmental variance PLoS One 3 e2777
Bishop M J et al Effects of ocean sprawl on ecological connectivity: impacts and solutions J. Exp. Mar. Biol. Ecol. accepted
Bowman R S and Lewis J R 1977 Annual Fluctuations in the recruitment of Patella vulgata L. Mar. Biol. Assoc. UK 57 793–815
Bracewell S A, Robinson L A, Firth L B and Knights A M 2013 Predicting free-space occupancy on novel artificial structures by an invasive intertidal barnacle using a removal experiment PLoS One 8 e74457
Brownie M A and Chapman M G 2011 Ecologically informed engineering reduces loss of intertidal biodiversity on artificial shorelines Environ. Sci. Technol. 45 8204–7
Brownie M A and Chapman M G 2014 Mitigating against the loss of species by adding artificial intertidal pools to existing seawalls Mar. Ecol. Prog. Ser. 497 119–29
Burt J A, Feary D A, Cavalcante G, Bauman G A and Usseglio P 2013 Urban breakwaters as reef habitat in the Persian gulf Mar. Poll. Bull. 72 342–50
Caffrey J M, Milline M, Evers S, Moron H and Butler M 2010 A novel approach to aquatic weed control and habitat restoration using biodegradable jute matting Aquat. Invas. 5 123–9
Chadwick M A, Dobberfuß D R, Benke A C, Huynh A D, Suberkropp K and Thiele J E 2006 Urbanization affects stream ecosystem function by altering hydrology, chemistry, and biotic richness Ecol. Appl. 16 1796–807
Chapman M G 2003 Paucity of mobile species on constructed seawalls: effects of urbanization on biodiversity Mar. Ecol. Prog. Ser. 264 21–9
Chapman M G and Blockley D J 2009 Engineering novel habitats on urban infrastructure to increase intertidal biodiversity Oecologia 161 625–35
Chapman M G and Underwood A J 2008 Scales of variation of gastropod densities over multiple spatial scales: comparison of common and rare species Mar. Ecol. Prog. Ser. 354 147–60
Chapman M G and Underwood A J 2011 Evaluation of ecological engineering of ‘armoured’ shorelines to improve their value as habitat J. Exp. Mar. Biol. Ecol. 400 302–13
Clarke K 1993 Non-parametric multivariate analyses of changes in community structure Austr. J. Ecol. 18 117–43
Clarke K R and Warwick R M 1994 PRIMER: Plymouth Routines in Multivariate Ecological Research, a Suite of Computer Programmes (Plymouth: Plymouth Marine Laboratory)
Coombes M A, La Marca E C, Naylor L A and Thompson R C 2015 Getting into the groove: opportunities to enhance the ecological value of hard coastal infrastructure using fine-scale surface textures Ecol. Eng. 77 314–23
Cunha A H, Marba N N, VanKatwijk M M, Pickering C, Henriques M, Bernard G, Ferreira M A, Garcia S, Garmendia J M and Manent P 2012 Changing paradigms in seagrass restoration Restor. Ecol. 20 427–30
Dafni K A, Glasby T M, Airolidi L, Rivera N K, Mayer-Pinto M and Johnston E I 2015 Marine urbanization: an ecological framework for designing multifunctional artificial structures Front. Ecol. Environ. 13 82–90
Dorazio R M, Gotelli N J and Ellison A M 2011 Modern methods of estimating biodiversity from presence–absence surveys Biodiversity Loss in a Changing Planet ed O Grillo and G Venora (Rijeka, Croatia: InTech) pp 277–302
Dyson K and Yokom K 2015 Ecological design for urban waterfronts Urban Ecosyst. 18 189–208
Edger M J and Murdock S 2003 Galway City Wastewater Treatment Plant: Design and Construction Aspects The Institution of Engineers of Ireland, Paper presented to the Institution of Engineers of Ireland Joint Civil and Water & Environmental Divisions, 7 April 2003 (www.engineersireland.ie/EngineersIreland/media/SiteMedia/groups/societies/ water-enviro/Galway_City_Wastewater_Treatment_Plant-_Design_and_Construction_Aspects.pdf) accepted
Elton C S 1958 The Ecology of Invasions By Animals and Plants (London: Methuen)
Eriksson B K, Johannsson G and Snoeijis P 2002 Long-term changes in the macroalgal vegetation of the inner Gullmar Fjord, Swedish Skagerrak coast J. Phycol. 38 284–96
Evans A J, Firth L B, Hawkins S J, Morris E S, Goudge H and Moore P J 2011a Drill-cored rock pools: an effective method of ecological enhancement on artificial structures Mar. FW. Res. 67 123–30
Evans A J, Garrod B, Firth L B, Hawkins S J, Morris E S, Goudge H and Moore P J 2016b Multi-functional coastal developments: desirability of potential secondary benefits and steps to effective implementation Marine Policy accepted
Ferrari F, Ivela L, Jaklin A, Perkol–Finkel S and Airolidi L J 2016 The overlooked role of biotic factors in controlling the ecological performance of artificial marine habitats J. Appl. Ecol. 53 16–24
Firth L B and Hawkins S J 2011 Global change in marine ecosystems: patterns, processes and interactions with regional and local scale impacts J. Exp. Mar. Biol. Ecol. 400 1–6
Firth L B, Knights A M and Bell S S 2011 Air temperature and winter mortality: implications for the persistence of the invasive mussel, Perna viridis in the intertidal zone of the south-eastern United States J. Exp. Mar. Biol. Ecol. 400 250–6
Firth L B, Knights A M, Thompson R C, Mieszkowska N, Bridger D, Evans A, Moore P J, O’Connor N E, Sheehan E V and Hawkins S J 2016 Ocean sprawl: challenges and opportunities for biodiversity management in a changing world Oceanogr. Mar. Biol. Annu. Rev. accepted
Firth L B, Mieszowska N, Thompson R C and Hawkins S J 2013a Climate change and adaptional impacts in coastal systems: the case of sea defences Environ. Sci. Proc. Impact. 15 1665–70
Firth L B, Schofield M, White F J, Skov M W and Hawkins S J 2014b Biodiversity in intertidal rock pools: informing engineering criteria for artificial habitat enhancement in the built environment Mar. Environ. Res. 102 122–30
Firth L B, Thompson R C, White F J, Schofield M, Skov M W, Hoggart S P G, Jackson J, Knights A M and Hawkins S J 2013b The importance of water-retaining features for biodiversity on artificial intertidal coastal defence structures Divers. Distrib. 19 1275–83
Firth L B et al 2014a Between a rock and a hard place: environmental and engineering considerations when designing coastal defence structures Coast. Eng. 87 122–35
Firth L B et al 2015 Historical comparisons reveal multiple drivers of decadal change of an ecosystem engineer at the range edge Ecol. Evol. 5 3210–22
Floril O, Inglis G J, Dey K and Smith A 2009 The importance of transport hubs in stepping-stone invasions J. Appl. Ecol. 46 37–45
García-Gómez J C, Guerra-García J M, Espinosa F, Maestre M J, Rivera-Ingraham G, Fa D, González A R, Ruiz-Tabares A and López-Fé C M 2015 Artificial marine micro-reserves: networks (AMMRN): an innovative approach to conserve marine littoral biodiversity and protect endangered species Mar. Ecol. 36 259–77
Gerland P et al 2014 World population stabilization unlikely this century Science 346 234–7
Griffin J N, Noël L M L J, Crowe T P, Burrows M T, Hawkins S J, Thompson R C and Jenkins S R 2010 Consumer effects on ecosystem functioning in rock pools: roles of species richness and composition Mar. Ecol. Prog. Ser. 420 43–56
Hawkins S J 1981 The influence of Patella grazing on the fucoid/barnacle mosaics on moderately exposed rocky shores Kieler Meeresforsch 5 537–43
Hawkins S J and Hartnell R G 1980 A study of the small-scale relationship between species number and area on a rocky shore Estuar. Coast. Mar. Sci. 10 201–14
Hawkins S J and Hartnell R G 1983 Grazing of intertidal algae by marine invertebrates Mar. Biol. Ann. Rev. 21 195–282
Hawkins S J, Southward A J and Barrett R L 1983 Population structure of Patella vulgata L. during succession on rocky shores in southwest England Oceanol. Acta SP 103–7
Hawkins S J et al 2013a Data rescue and reuse: recycling old information for new concerns Mar. Pol. 42 91–8
Hawkins S J, Vale M, Firth L B, Burrows M T, Mieszowska N and Frost M 2013b Sustained observation of marine biodiversity and ecosystems Est. Coast. Shelf Sci. 89–90 1101–10
Huntingford C et al 2014 Potential influences on the United Kingdom’s floods of winter 2013/14 Nat. Clim. Change 4 769–77
Inger R et al 2009 Marine renewable energy: potential benefits to biodiversity? An urgent call for research J. Appl. Ecol. 46 1145–53
Johnson M P 2001 Environmental impacts of urban sprawl: a survey of the literature and proposed research agenda Environ. Plan. A 33 A 717–35
Jonsson P R, Granhag L, Moschella P S, Åberg P, Hawkins S J and Thompson R C 2006 Interactions between wave action and grazing control the distribution of intertidal macroalgae Ecology 87 1169–78
Knights A M et al 2015 An exposure-effect approach for evaluating ecosystem-wide risks from human activities ICES J. Mar. Sci. 72 1105–15
Knights A M, Firth L B, Thompson R C, Yunnie A L, Hiscock K and Hawkins S J 2016 Plymouth—a world harbour through the ages Reg. Stud. Mar. Sci. (doi:10.1016/j.rsmas.2016.02.002)
Lai S, Loke L H L, Hilton M J, Bouma T J and Todd P A 2015 The effects of urbanisation on coastal habitats and the potential for ecological engineering: a Singapore case study Ocean Coast. Manage. 103 76–85
Lirman D and Miller M W 2003 Modeling and monitoring tools to assess recovery status and convergence rates between restored and undisturbed coral reef habitats Restor. Ecol. 11 448–56
Loke L H L and Todd P A 2016 Structural complexity and component type increase intertidal biodiversity independently of area Ecology 97 383–93
Martins G M, Thompson R C, Neto A I, Hawkins S J and Jenkins S R 2010 The effects of the exploited limpet Patella candei d’orbigny via modifications in coastal engineering Biol. Conserv. 143 203–11
Matias M G, Chapman M G, Underwood A J and O’Connor N E 2012 Increasing density of rare species of intertidal gastropods: tests of competitive ability compared with common species Mar. Ecol. Prog. Ser. 453 107–16
Matthews T, Murphy C, Wilby R L and Harrigan S 2014 Stormiest winter on record for Ireland and UK Nat. Clim. Change 4 73–80
Mieszowska N, Sugden H, Firth L B and Hawkins S J 2014 The role of sustained observations in tracking impacts of environmental change on marine biodiversity and ecosystems Phil. Trans. R. Soc. A 372 20130339
Mineur F, Cook E J, Minchin D, Bohn K, Macleod A and Maggs C A 2012 Changing coasts: marine aliens and artificial structures Oceanogr. Mar. Biol. Ann. Rev. 50 189–234
Moore P, Thompson R C and Hawkins S J 2007 Effects of grazer identity on the probability of escapes by a canopy-forming macroalga J. Exp. Mar. Biol. Ecol. 344 170–80
Moschella P S et al 2005 Low-crest coastal defence structures as artificial habitats for marine life: using ecological criteria in design Coast. Eng. 52 1053–71
Mrowicki R J, Maggs C A and O’Connor N E 2015 Consistent effects of consumer species loss across different habitats Oikos 124 355–65
Mrowicki R J, O’Connor N E and Donohue I 2015 Temporal variability of a single population can determine the vulnerability of communities to perturbations J. Ecol. in press (doi:10.1111/1365-2745.12533)
Nandakumar K, Matsunaga H and Takagi M 2003 Microfouling studies on experimental test blocks of steel-making slag and concrete exposed to seawater off Chiba, Japan Biofouling 19 257–67
Ng C S L, Lim S C, Ong J Y, Teo L M S, Chou L M, Chua K E and Tan K S 2015 Enhancing the biodiversity of coastal defence structures: transplantation of nursery-reared reef biota onto intertidal seawalls Ecol. Eng. 82 480–6
Noël L M L J, Griffin J N, Thompson R C, Hawkins S J, Burrows M T, Crowe T P and Jenkins S R 2010 Assessment of a field incubation method estimating primary productivity in rockpools and communities Est. Coast. Shelf Sci. 88 155–9
O’Connor N E and Crowe T P 2005 Biodiversity loss and ecosystem functioning: distinguishing between number and identity of species Ecology 86 1783–96
Oksanen J et al 2016 Vegan: Community Ecology Package. R package version 2.3–4
Pearce B, Farías-Franco J M, Wilson C, Pitts J and Somerfield P J 2014 Repeated mapping of reefs constructed by Sabellaria spinulosa Leuckart 1849 at an offshore wind farm site Contin. Shelf Res. 83 3–13
Perkol-Finkel S, Ferrario F, Nicotera V and Airoldi L 2012 Conservation challenges in urban seascapes: promoting the growth of threatened species on coastal infrastructures J. Appl. Ecol. 49 1457–66
Perkol-Finkel S and Sella I 2014 Ecologically active concrete for coastal and marine infrastructure: innovative matrices and designs From Sea to Shore—Meeting the Challenges of the Sealed W Allsop and K Burgess (London, UK: ICE) pp 1139–49
Perkol-Finkel S and Sella I 2015 Harnessing urban coastal infrastructure for ecological enhancement Proc. ICE-Marine Eng. 168 102–10
Pister B 2009 Urban marine ecology in southern California: the ability of riprap structures to serve as rocky intertidal habitat Mar. Biol. 156 661–73
Raffaelli D and Hawkins S 1996 *Intertidal Ecology* (London: Chapman and Hall)
Risinger J D 2012 Biologically dominated engineered coastal breakwaters PhD Thesis Louisiana State University and Agricultural and Mechanical College
Sella I and Perkol-Finkel S 2015 Blue is the new green—ecological enhancement of concrete based coastal and marine infrastructure *Ecol. Eng.* 84 260–72
Sharma P 2009 *Coastal Zone Management* (New Delhi: Global India)
Simkanin C, Dower J F, Filip N, Jamieson G and Therriault T W 2013 Biotic resistance to the infiltration of natural benthic habitats: examining the role of predation in the distribution of the invasive ascidian *Botrylloides violaceus* *J. Exp. Mar. Biol. Ecol.* 439 76–83
Skov M W, Hawkins S J, Volkelt-Igoe M, Pike J, Thompson R C and Doncaster C P 2011 Patchiness in resource distribution mitigates habitat loss: insights from high-shore grazers *Ecosphere* 2 1160
Sousa W P 1979 Experimental investigations of disturbance and ecological succession in a rocky intertidal algal community *Ecol. Monogr.* 49 227–54
Southward A J and Southward E C 1978 Recolonization of rocky shores in Cornwall after use of toxic dispersants to clean up the torrey canyon spill *J. Fish. Board Canada* 35 682–706
Stachowicz J J, Fried H, Orman R W and Whittaker R B 2002 Biodiversity, invasion resistance, and marine ecosystem function: reconciling pattern and process *Ecology* 83 2575–90
Thompson R C, Crowe T P and Hawkins S J 2002 Rocky intertidal communities: past environmental changes, present status and predictions for the next 25 years *Environ. Conserv.* 29 168–91
Toff J D, Ogston A S, Heerhartz S M, Cordell J R and Flemer E E 2013 Ecological response and physical stability of habitat enhancements along an urban armored shoreline *Ecol. Eng.* 57 97–108
Tuomisto H 2010a A diversity of beta diversities: straightening up a concept gone awry: I. Defining beta diversity as a function of alpha and gamma diversity *Ecography* 33 2–22
Tuomisto H 2010b A diversity of beta diversities: straightening up a concept gone awry: II. Quantifying beta diversity and related phenomena *Ecography* 33 23–45
Underwood A J 1997 Ecological experiments: their logical design and interpretation using analysis of variance (Melbourne, Australia: Cambridge University Press)
Van Metre P C, Mahler B J and Furlong E T 2000 Urban sprawl leaves its PAH signature *Environ. Sci. Technol.* 34.19 4064–70
Whittaker R H 1960 Vegetation of siskiyou mountains, Oregon and California *Ecol. Monogr.* 30 279–338
Wyles K J, Pahl S and Thompson R C 2014 Perceived risks and benefits of recreational visits to the marine environment: integrating impacts on the environment and impacts on the visitor *Ocean Coast. Manage.* 88 53–63