

Drill-cored rock pools: an effective method of ecological enhancement on artificial structures

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Abstract. Coastal defences are proliferating in response to anticipated climate change and there is increasing need for ecologically sensitive design in their construction. Typically, these structures support lower biodiversity than natural rocky shores. Although several studies have tested habitat enhancement interventions that incorporate novel water-retaining features into coastal defences, there remains a need for additional long-term, fully replicated trials to identify alternative cost-effective designs. We created artificial rock pools of two depths (12 cm, 5 cm) by drill-coring into a shore-parallel intertidal granite breakwater, to investigate their potential as an intervention for delivering ecological enhancement. After 18 months the artificial rock pools supported greater species richness than adjacent granite rock surfaces on the breakwater, and similar species richness to natural rock pools on nearby rocky shores. Community composition was, however, different between artificial and natural pools. The depth of artificial rock pools did not affect richness or community structure. Although the novel habitats did not support the same communities as natural rock pools, they clearly provided important habitat for several species that were otherwise absent at mid-shore height on the breakwater. These findings reveal the potential of drill-cored rock pools as an affordable and easily replicated means of enhancing biodiversity on a variety of coastal defence structures, both at the design stage and retrospectively.

Additional keywords: coastal protection, complexity, conservation, ecological engineering, management, urban ecology.

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Introduction

Globally, engineered structures such as breakwaters, groynes and seawalls are proliferating in the marine environment, as a societal adaptational response to anticipated climate change (IPCC 2013), and to protect expanding coastal developments (UN Atlas of the Oceans: UN 2014). In parts of Europe (Airoldi and Beck 2007), the USA (Davis *et al.* 2002) and Australia (Chapman and Bulleri 2003) over half of natural shorelines have been replaced or reinforced by artificial hard substrata. The primary objective of most coastal defence developments is to protect assets from flooding and erosion. They will, however, inevitably be colonised by epibenthic marine organisms, thereby having the potential to fulfil secondary management objectives, such as enhancement of biodiversity or

provision of habitat for target species (see Firth *et al.* 2013a for review).

It has been well documented that intertidal coastal defences typically support organisms normally found on nearby rocky shores (Chapman 2003; Pinn *et al.* 2005; Moschella *et al.* 2005; Firth *et al.* 2015, 2016), but that the colonising communities are often not completely analogous. In particular, there is growing evidence that artificial structures support lower biodiversity (Chapman 2003; Moschella *et al.* 2005; Pister 2009; Firth *et al.* 2013b) and different relative abundances of taxa (Chapman and Bulleri 2003; Pinn *et al.* 2005; Moschella *et al.* 2005) compared with adjacent rocky shores. The diversity deficits reported on artificial coastal structures have been attributed to a variety of organisms, including mobile fauna

(Chapman 2003; Pister 2009), lower-shore and other desiccation-sensitive taxa (Moschella *et al.* 2005; Firth *et al.* 2016), and proportionally rarer taxa (Chapman 2003).

It has been suggested that the lower epibiotic diversity recorded on these structures, compared with equivalent natural habitats, is partly due to higher disturbance regimes caused by wave energy and sand scouring (Moschella *et al.* 2005). Coastal defences are often constructed in high-energy, erosive soft-sediment environments, where there tends to be greatest need for coastal protection. In addition, intermittent maintenance activities may also cause considerable disturbance to ecological communities (Airoldi and Bulleri 2011). Low biodiversity has also been linked to limited habitat extent (Moschella *et al.* 2005; Chapman and Underwood 2011), increased intertidal gradient (Chapman 2003; Chapman and Underwood 2011) and reduced topographic complexity (Chapman 2003; Moschella *et al.* 2005), when compared with natural rocky shores. At the finest scale of complexity (<1 cm), coastal defence structures are often built using materials with smoother surface texture than natural rock (e.g. quarried granite or concrete). At small (1–100s cm) and medium (1–100s m) scales, they tend to be reasonably homogeneous in terms of habitat shape and structure. Moschella *et al.* (2005) suggested that species diversity may be increased on artificial structures through engineering interventions and that the scope for such interventions is greatest at the microhabitat scale (1–100s cm). Subsequent experimental trials recommended that settlement and survival could be maximised by incorporating multiple novel habitats with a variety of depths and diameters (from the 10-m to <1-cm scale: Firth *et al.* 2014a). As a consequence, there is currently much interest in developing novel designs for coastal defence structures that incorporate more heterogeneous microhabitats.

Small pits and crevices are important microhabitats for rocky shore biota, providing shade, moisture and refuge from predation and disturbance (Fairweather 1988; Gray and Hodgson 1998; Johnson *et al.* 1998). The addition of pits and crevices to artificial structures can be an effective way of increasing species richness (Firth *et al.* 2014a) and enhancing stocks of exploited species (Martins *et al.* 2010). However, the addition of microhabitat complexity at this small scale may have a limited long-term effect on biodiversity if the pits and crevices are rapidly occupied by already-abundant mobile fauna or larval settlement events, and thus are unavailable for colonisers that arrive later in the successional trajectory (Chapman and Underwood 2011). Browne and Chapman (2014) therefore suggested that engineering interventions modelled on small- and medium-sized rock pools may be more likely to support persisting trends in increased biodiversity. Rock pools provide refuge from fluctuations in temperature and desiccation stress and can extend the vertical distribution of lower-shore species (see Metaxas and Scheibling 1993 for review). It has recently been demonstrated that rock pools support greater species diversity than emergent substrata in both natural (Firth *et al.* 2013b, 2014b) and artificial (Firth *et al.* 2013b) habitats.

Through collaborations with engineers, water-retaining features mimicking rock pools were added to vertical seawalls in Sydney Harbour, both during construction (i.e. shaded recesses with water-retaining lips: Chapman and Blockley 2009) and retrospectively (i.e. concrete flower pots bracketed to walls: Browne and Chapman 2014). These engineered habitats were

colonised by a variety of intertidal organisms and were found to be easy, cost-effective ways of enhancing the ecological condition of vertical seawalls. However, both designs had limitations: the shaded recesses may not provide suitable habitat for the full range of intertidal taxa because they are continually shaded; and several of the flower pots were lost from the walls within 7 months of deployment. It is also unlikely that these novel habitats would perform in the same way on other types of artificial structures in other locations. Habitat enhancement interventions intended for coastal defence schemes should be robust against anticipated disturbance from extreme weather events in exposed environments. Sydney Harbour is a semi-enclosed environment with a small tidal range, unlike the exposed open shorelines where coastal defence structures are frequently required. Therefore, there remains a need for additional long-term, fully replicated trials to determine the potential biodiversity benefits of incorporating different types of water-retaining features within coastal defence structures.

Here we investigate novel drill-cored artificial rock pools as a durable, affordable and replicable habitat enhancement intervention on an intertidal coastal defence breakwater. We evaluate their potential to increase biodiversity on the breakwater and to provide surrogate habitat for rocky shore communities. The cost of this intervention is directly related to the time taken to drill the artificial pools, which in turn is directly related to the depth of the pools created. Therefore, although the artificial pools were all designed to replicate small unshaded rock pools on nearby rocky shores, we trialled the design with two different depths (12 cm and 5 cm). Martins *et al.* (2007) suggested that deeper rock pools support greater species diversity (and different communities) than shallow ones because they provide more stable environmental conditions (but see Firth *et al.* 2014b). In natural systems, there is little distinction in the habitat provided by 5 and 12 cm-deep rock pools, and previous studies consider all pools in this range to be shallow (e.g. Firth *et al.* 2014b). However, the effect of depth in artificial systems is not well understood (but see Browne and Chapman 2014) and ecological responses may be more pronounced (Firth *et al.* 2013b).

We compared biodiversity in the drill-cored artificial rock pools with adjacent granite rock surfaces on the breakwater, testing hypotheses that: (1) the artificial rock pools would support greater species richness than adjacent emergent rock surfaces; and (2) deeper (12 cm) artificial rock pools would support greater species richness than, and different community structure to, shallower (5 cm) ones. To assess their potential as surrogate habitats for rocky shore communities, we also compared the artificial rock pools with natural rock pools, testing a third hypothesis: (3) the artificial rock pools would support equivalent species richness and community structure to natural rock pools on nearby rocky shores. Finally, we evaluate the management implications of this engineering intervention as a habitat enhancement tool for new and existing coastal defences that are becoming ubiquitous features of urban coastlines globally.

Materials and methods

Study area

Novel water-retaining habitats (artificial rock pools) were added to an intertidal granite riprap breakwater at Tywyn, West Wales,

UK (52°34.8'N, 04°05.9'W). The breakwater was constructed in 2010 and is positioned on a sandy shore, backed by a concrete sea-wall. The breakwater was compared with three nearby natural rocky shore sites: Aberystwyth (52°25.1'N, 04°05.2'W), Borth (52°28.8'N, 04°03.3'W) and Clarach (52°26.2'N, 04°04.9'W). All are moderately exposed bedrock shores of mixed sandstones and mudstones, with shallow gradients and sandy surroundings similar to Tywyn. The three natural rocky shore sites were located to the south of the breakwater due to a lack of suitable sites to the north. Although this was not optimal for the purposes of experimental design, the species pool to the north (Llyn Peninsula) and south (natural reefs used in this study) is the same (MarClim data: S. J. Hawkins, M. T. Burrows, L. B. Firth, M. J. Genner, R. J. H. Herbert, S. R. Jenkins, P. J. Moore, E. Poloczanska, A. J. Southward, H. Sugden, R. C. Thompson and N. Mieszkowska, unpubl. data). Local coastal processes are characterised by predominantly wave-driven northerly drift of material and highly dynamic beach profiling.

Experimental plots

Eighteen artificial rock pools were drilled into the horizontal granite surfaces at mid-shore height on the seaward side of the breakwater, using a diamond-tipped drill corer (Fig. 1). The pools were designed to replicate small unshaded natural rock pools on nearby rocky shores; however, to make them affordable and easily replicable, they were necessarily more uniform than natural rock pools (Metaxas and Scheibling 1993). They were all cylindrical with 15-cm diameter and were of two different depths ('deep' 12 cm, 'shallow' 5 cm), randomly assigned with nine replicates of each. Eighteen permanent quadrats were marked out on emergent freely draining rock surfaces surrounding the drilled pools. Quadrats were of two different sizes, equal to the surface area of the 'deep' and 'shallow' pools (i.e. 'deep' 742 cm², 'shallow' 413 cm², calculated as the combined surface area of the bottoms and sides of the cylindrical pools), also randomly assigned with nine replicates of each.

At each of the three natural rocky shores, 10 natural rock pools were selected at mid-shore height for comparison. Natural

pools were selected to have comparable dimensions to the artificial pools, i.e. ~15-cm diameter and between five and 12 cm deep. There was no significant difference in species richness or community structure between the 'deep' and 'shallow' artificial rock pools. We therefore did not account for the depth treatment when comparing artificial with natural pools, but used a subset of 10 artificial replicates (five 12 cm, five 5 cm, randomly selected), pooled over depth to increase statistical power in analyses.

In April 2012 all biota were scraped from the experimental plots (i.e. artificial rock pools, emergent surfaces and natural rock pools), then each plot was treated with a flame-gun to remove biofilms and calcareous deposits. A radius of 50 cm around each plot was also cleared and heat-treated to avoid vegetative recolonisation. This was defined as T_{zero} , at which point the artificial rock pools, emergent rock surfaces and natural rock pools began their trajectories of colonisation simultaneously. The habitats were monitored over 18 months (monthly for the first 3 months and quarterly thereafter) to record counts of mobile fauna and percentage cover of sessile fauna and algae. Taxa were recorded to species level, but where this was not possible without destructive sampling, consistent morphospecies were used, e.g. *Lithothamnium*.

Data analysis

To address our first hypothesis that the artificial rock pools would support greater species richness than emergent rock surfaces, total richness and species accumulation (using presence-absence) were pooled over replicates ($n = 18$) and plotted over time (18 months). Analysis of variance (ANOVA) was used to test for differences in mean species richness (after 18 months) between the two habitats. A two-way crossed design was used, with fixed factors Habitat (two levels: pool, surface) and Depth (two levels: 'deep', 'shallow'), and $n = 9$.

To address our second hypothesis that deeper (12 cm) artificial rock pools would support greater species richness than, and different community structure to, shallower (5 cm) pools, ANOVA was used to test for differences in mean species richness (after 18 months) between the two habitats. A simple one-way design was used, with the single fixed factor Depth (2 levels: 'deep', 'shallow') and $n = 9$. Permutational analysis of variance (PERMANOVA: Anderson 2001) was used to test for differences in multivariate species assemblages (after 18 months), based on 9999 unrestricted permutations of raw data.

To address our third hypothesis that the artificial rock pools would support equivalent species richness and community structure to natural rock pools, a subset of 10 artificial rock pools (five 12 cm, five 5 cm, randomly selected and pooled over depth) were compared with 10 natural pools at each of the natural rocky shores. ANOVA and PERMANOVA (with 9999 permutations of residuals under a reduced model) were used to test for differences in mean species richness and multivariate species assemblages (after 18 months) respectively. For both analyses, a two-way asymmetrical nested design was used, with the random factor Site (four levels: Tywyn, Aberystwyth, Borth, Clarach) nested within the fixed upper-level factor Habitat (two levels: artificial, natural), with $n = 10$. PERMANOVA analyses were performed on the full communities and on the mobile and sessile components separately. Percentage contributions of individual species to



Fig. 1. Artificial rock pools (diameter: 15 cm, depth: 12 cm, 5 cm) drilled into the horizontal surfaces of an intertidal granite breakwater at Tywyn, UK (52°34.8'N, 04°05.9'W).

dissimilarities between habitat communities (full communities) were calculated using the SIMPER routine (Clarke 1993).

Univariate analyses were carried out in SPSS ver. 21 (IBM Corp., Armonk, NY, USA). Prior to analyses, homogeneity of variance was confirmed using Levene's test. Data comparing artificial pools and emergent rock surfaces did not meet this assumption, even following transformation. As ANOVA is robust to the departure from this assumption with balanced designs (Underwood 1997), the analysis was performed. However, results at the $P < 0.05$ significance level should be interpreted with some caution. Multivariate analyses were carried out in PRIMER ver. 6 and PERMANOVA+ ver. 6 (PRIMER-E Ltd, Plymouth, UK) and were based on Bray–Curtis similarity matrices of fourth-root transformed data. Homogeneity of multivariate dispersions was confirmed before multivariate analyses using the PERMDISP routine. For the asymmetrical PERMANOVA analysis there were not enough possible permutations to perform a reasonable test of significance, therefore Monte Carlo P values were used as a more meaningful, but approximate, alternative (Anderson and Robinson 2003).

Results

Comparing artificial pools with emergent surfaces on the breakwater

Collectively, the artificial rock habitats consistently supported more species than the surrounding emergent rock surfaces (Fig. 2a). Initially, total species richness increased rapidly, reaching a maximum after 3 months on the emergent rock surfaces (13 species) and after 6 months in the artificial pools (22 species). Although total richness appeared to then level off (but with some seasonal fluctuations), species accumulation curves (Fig. 2a) revealed that the total species pool supported by both habitats did, in fact, continue to rise steadily over time. Species accumulation on the emergent rock surfaces, however, appears to have reached an asymptote after 9 months, while for the pools an asymptote had not yet been reached (Fig. 2a). This suggests that the artificial rock pools not only supported greater diversity (species richness) overall, but also a greater diversity of transient and ephemeral taxa, which were utilising the novel habitats at different times of year.

After 18 months, the mean species richness in the artificial pools ('deep': 8.3 ± 1.4 s.e., 'shallow': 8.1 ± 2.6 s.e.) was significantly greater than on the adjacent rock surfaces ('deep': 6.1 ± 1.5 s.e., 'shallow': 5.7 ± 1.6 s.e.) ($F_{1,32} = 6.380, P = 0.017$) (Table S1). There was no significant effect of depth ($F_{1,32} = 0.130, P = 0.721$) and no interaction ($F_{1,32} = 0.014, P = 0.905$). The rock pools increased the richness of most taxonomic groups (with the exception of mobile crustaceans, barnacles and bivalves) and provided habitat for several major groups that were absent from the surrounding granite rock surfaces (i.e. fish, ascidians, bryozoans, hydroids and sponges) (Fig. 2b).

Comparing deep and shallow artificial rock pools

There was no significant difference in species richness ($F_{1,16} = 0.017, P = 0.898$) (Table S2) or community structure (pseudo- $F_{1,16} = 1.240, P(\text{perm}) = 0.285$) (Table S3) between 'deep' (12 cm) and 'shallow' (5 cm) artificial rock pools after 18 months.

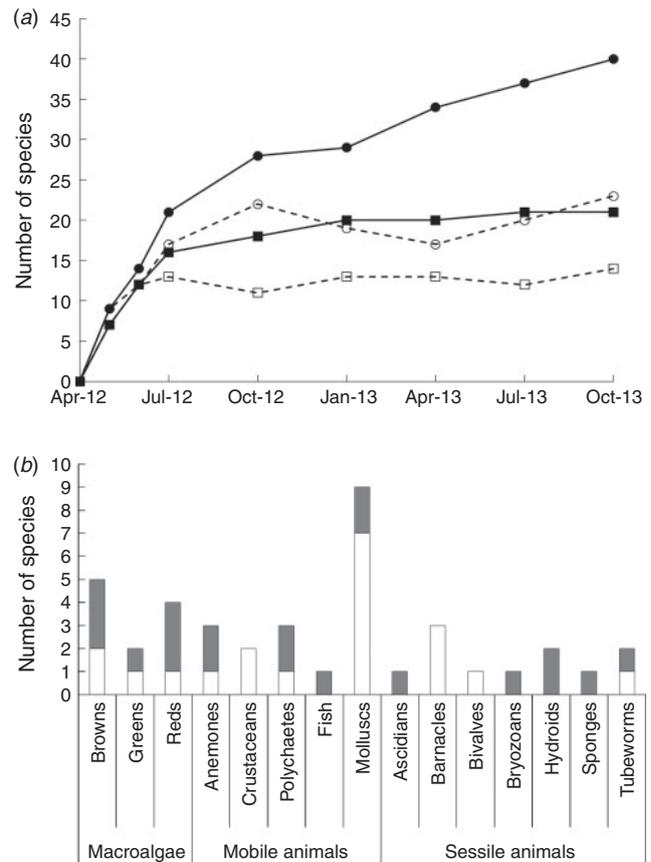


Fig. 2. (a) Total species richness (dashed lines) and cumulative number of species (solid lines) recorded in artificial rock pools (circles) and on emergent rock surfaces (squares) over 18 months; (b) Total number of species in major taxa recorded on emergent rock surfaces (white bars) and additional species recorded exclusively in artificial rock pools (grey bars) after 18 months. Data pooled over 18 replicates in each case.

Comparing artificial pools with natural rock pools

After 18 months, there was no significant difference between mean species richness in the artificial rock pools and the natural rock pools ($F_{1,36} = 0.041, P = 0.858$) (Table S4). Community structure was, however, significantly different (pseudo- $F_{1,36} = 5.478, P(\text{mc}) = 0.016$) (Fig. 3a; Table S5). The dissimilarity between artificial pools and natural pools was attributable to sessile encrusting fauna and algae (pseudo- $F_{1,36} = 5.962, P(\text{mc}) = 0.027$) (Fig. 3b; Table S5), but not mobile and sedentary fauna (pseudo- $F_{1,36} = 2.194, P(\text{mc}) = 0.150$) (Fig. 3c; Table S5), which included several species of molluscs, crustaceans and anemones. SIMPER analysis (Table 1) reported that almost 50% of the dissimilarity observed between the artificial and natural rock pool communities was attributed to two calcareous algae, *Corallina officinalis* (7.8% contribution) and encrusting *Lithothamnium* (9.6%), the ephemeral green alga *Ulva intestinalis* (5.1%), and to sessile fauna such as *Sabellaria alveolata* (6.0%), *Mytilus edulis* (11.7%) and *Austrominius modestus* (6.4%). Each of these was more abundant in the artificial rock pools than in the natural pools, with the exception of the two coralline algal species, which were present in natural pools but absent from the artificial ones.

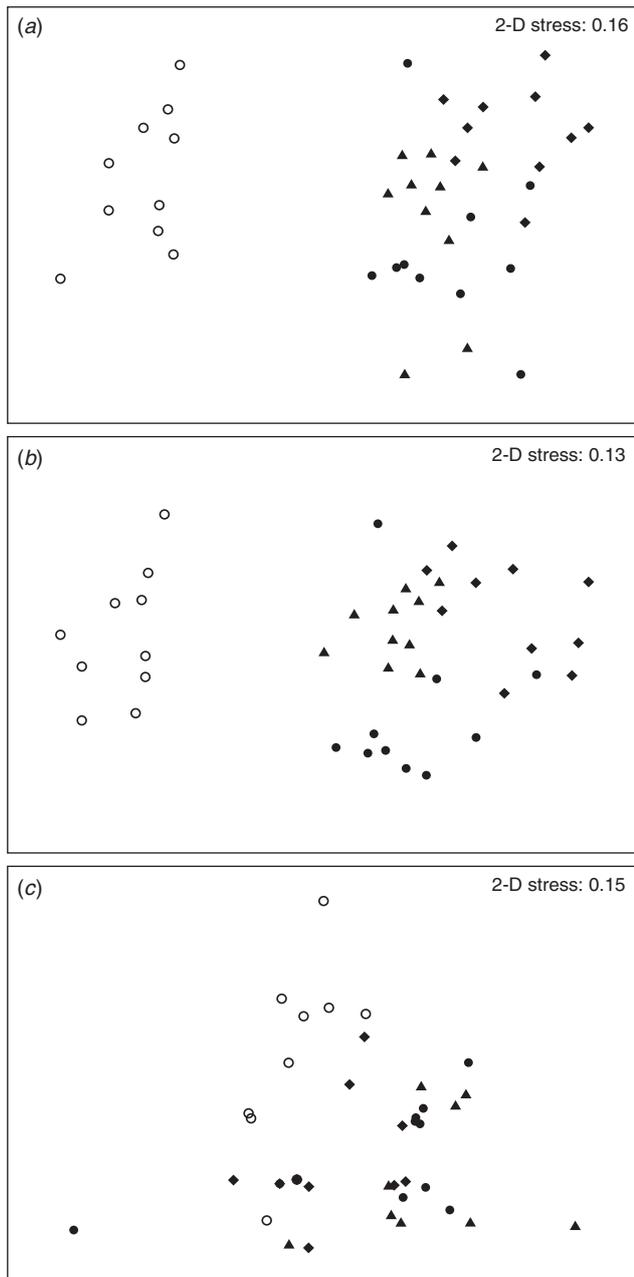


Fig. 3. nMDS ordinations of multivariate species assemblages after 18 months: (a) full community; (b) sessile species; and (c) mobile species. Open circles represent artificial rock pools at Tywyn and filled symbols represent natural rock pools at Aberystwyth (triangles), Borth (circles) and Clarach (diamonds).

Discussion

Factors influencing biodiversity

The addition of drill-cored artificial rock pools increased the number of species living on the mid-shore breakwater units studied. The pools were utilised by many species that were absent on the adjacent emergent rock surfaces. Further, they supported a much larger species pool than the emergent surfaces over time, and a comparable number of species to natural rock

Table 1. Differences (< and >) in mean abundances (counts (c) or percentage cover (%)) of species recorded in artificial rock pools ($n = 10$) and natural rock pools ($n = 30$) after 18 months

Species are listed in order of their contribution (%) to the dissimilarities between multivariate species assemblages (SIMPER analysis on full community). %, percentage contribution to multivariate dissimilarity; Diss./s.d., dissimilarity divided by standard deviation of contributions across all pairs of samples (this is a measure of consistency of contribution across replicates). Average dissimilarity = 76.3%

Species	Artificial pools		Natural pools	%	Diss./s.d.
<i>Mytilus edulis</i> %	29.40	>	0.03	11.70	3.09
<i>Lithothamnium</i> %		<	16.30	9.58	2.66
<i>Corallina officinalis</i> %		<	6.47	7.77	4.24
<i>Austrominius modestus</i> %	5.10	>	0.05	6.42	1.69
<i>Sabellaria alveolata</i> %	5.15	>	0.67	6.03	1.67
<i>Ulva intestinalis</i> %	8.30	>	1.33	5.09	1.23
<i>Littorina littorea</i> c	2.70	>	0.83	4.27	1.09
<i>Actinia equina</i> c	1.00	>		4.03	1.38
<i>Gibbula umbilicalis</i> c		<	1.93	4.00	1.26
<i>Ceramium</i> spp. %	1.65	>	0.52	3.80	1.13
<i>Polysiphonia</i> spp. %	1.85	>	0.93	3.75	1.15
<i>Osmundea osmundea</i> %		<	1.22	3.74	1.26
<i>Fucus vesiculosus</i> %	1.70	>	0.95	3.32	0.99
<i>Semibalanus balanoides</i> %	0.50	<	1.08	3.14	1.03
<i>Patella vulgata</i> c	3.60	<	6.13	2.86	0.85
<i>Phumulariidae</i> sp. %	2.80	>		2.71	0.63
<i>Nucella lapillus</i> c	0.60	>	0.03	2.62	0.77
<i>Spirobranchus triqueter</i> %	1.70	>	0.60	2.57	0.71
<i>Chondrus crispus</i> %		<	1.23	2.48	0.79
<i>Campanulariidae</i> sp. %	0.20	<	1.37	2.30	0.67
<i>Chthamalus montagui</i> %		<	0.23	1.48	0.58
<i>Lipophrys pholis</i> c	0.20	>	0.13	1.47	0.6
<i>Anurida maritima</i> c		<	0.57	0.76	0.32
<i>Littorina obtusata</i> c	0.10	>	0.03	0.63	0.38
<i>Sagartia troglodytes</i> c	0.10	>		0.57	0.33
<i>Patella depressa</i> c		<	0.13	0.54	0.33
<i>Actinia fragacea</i> c	0.10	>		0.49	0.33
<i>Molgula</i> sp. c	0.10	>		0.49	0.33
Porifera crust (orange) %		<	0.12	0.40	0.26
<i>Chorda filum</i> %		<	0.13	0.32	0.27
<i>Electra pilosa</i> %		<	0.07	0.20	0.18
<i>Carcinus maenas</i> c		<	0.03	0.15	0.18
<i>Phorcus lineatus</i> c		<	0.03	0.16	0.18
Ascidian sp. c		<	0.03	0.14	0.18

pools on nearby rocky shores. Community structure recorded in the artificial pools, however, was different from that recorded in the natural pools, on account of differences in sessile assemblages, i.e. algae and encrusting fauna.

It has been reported that lower-shore and other desiccation-sensitive taxa are often absent from coastal defences (Moschella *et al.* 2005; Firth *et al.* 2016). Some of the taxa that were recorded exclusively in the artificial pools during this study were known to be present at lower-shore heights at the base of the breakwater units, where some moisture is retained at low tide (e.g. some members of the hydroids, ascidians and anemones: A. J. Evans, unpubl. data). It therefore appears that the artificial pools performed a similar function as rock pools in natural habitats (Metaxas and Scheibling 1993), by enabling extension of the

vertical distribution of some species to mid-shore level on the structure. This may increase the capacity of the breakwater to support viable populations of these taxa, because the steep profile and high disturbance regime characteristic around the base of coastal defences (Moschella *et al.* 2005) may limit the extent of available habitat and reduce survivorship at low-shore heights.

Previous research has also attributed low biodiversity on coastal defences to a paucity of mobile fauna and proportionally rarer taxa (Chapman 2003; Pister 2009). The artificial rock pools in this study appear to address both of these diversity deficits to some extent. Many of the species recorded exclusively in the artificial pools (e.g. sponges, bryozoans and red algae) were present in much lower abundances overall than the species that were common to both habitats (e.g. barnacles, gastropods and mussels). The pools also supported several species of mobile animals that were not otherwise living on the emergent rock surfaces. Furthermore, the mobile species assemblages in the artificial pools were equivalent to those found in natural rock pools in the surrounding area. The same was not true, however, for the sessile components of the communities, which were significantly different in the artificial pools compared to natural pools.

In general, the artificial rock pools supported more opportunistic algae (*U. intestinalis*) and sessile animals (*S. alveolata*, *M. edulis* and *A. modestus*) than the natural pools, but coralline algae (erect *C. officinalis* and encrusting Lithothamnina) were notably absent. Because the mobile faunal assemblages were the same in both habitats, it is unlikely that herbivory (e.g. O'Connor and Crowe 2008) or predation (e.g. Johnson *et al.* 1998) were dominant factors controlling sessile assemblage development. Instead, it is likely that physical and propagule supply differences between the breakwater and natural rocky shore environments were more influential. The artificial rock pools were cored into granite rock, thus their surfaces lacked the complexity and roughness of local natural intertidal rock (mixed sandstones and mudstones). Further, they were built into a breakwater of limited extent, surrounded by sandy sediment, which meant they were subject to increased scouring and were further from natural rocky shore source populations. Although the breakwater supported many common rocky shore organisms, several taxa were absent (A. J. Evans, unpubl. data); the nearest source populations for those missing taxa were at a distance of >1000 m, compared to <10 m on natural shores.

Opportunistic ephemeral species, such as *Ulva* spp., can be reliably predicted to colonise bare substrata (Sousa 1979) but may be outcompeted by perennial species in more benign environments (Martins *et al.* 2007). This may explain the higher abundances of *U. intestinalis* in the more-disturbed artificial pools, and higher abundances of perennial corallines in the natural pools. Encrusting algae and some turf species are, however, considered reasonably stress-tolerant (Steneck and Dethier 1994), and although the smooth surface texture of the artificial rock pools may not have been optimal for propagule settlement, *C. officinalis* is capable of colonising smooth surfaces (Harlin and Lindbergh 1977) and is ubiquitous on granite and artificial boulder shores elsewhere (e.g. Pister 2009). It is likely, therefore, that the absence of coralline algae in the artificial pools was due to limited dispersal capability (Dethier *et al.* 2003) because it was absent from the surrounding breakwater units (A. J. Evans, unpubl. data). Conversely, for

S. alveolata, *M. edulis* and *A. modestus*, source populations were locally available at all sites (A. J. Evans, pers. obs.). Therefore, although known gregarious settlers (Knight-Jones and Stevenson 1950; Wilson 1968; Seed 1969), distance to source population was not a limiting factor for these species in the natural pools. It is possible that the tube-building polychaete, *S. alveolata*, was more abundant in the artificial pools because of favourable hydrodynamics and a plentiful supply of suspended sand particles from the surrounding habitat (Firth *et al.* 2015). The distribution of *M. edulis* on open coasts has historically been considered erratic and unpredictable (Seed 1969), whereas *A. modestus* is invariably observed on artificial substrata (e.g. Volckaert *et al.* 2002; Bracewell *et al.* 2013) and may have a competitive advantage when colonising bare surfaces because of its ability to reproduce throughout much of the year (Muxagata *et al.* 2004). Coralline algae, *S. alveolata* and *M. edulis* can all be dominant space-occupiers (Suchanek 1978; Cunningham *et al.* 1984; Connell and Glasby 1999) and are important ecosystem engineers, encouraging settlement (Barnes and Gonor 1973) and providing habitat for other organisms (Seed 1996; Kelaher *et al.* 2001; Dubois and Retie 2002). It is therefore likely that the different relative abundances of these taxa may have had consequential implications for overall community development in artificial and natural rock pools, leading to the significant differences observed.

It is generally accepted that shallower rock pools experience more extreme environmental conditions than deeper pools (Metaxas and Scheibling 1993) and may also support lower diversity (Martins *et al.* 2007; but see Firth *et al.* 2014b). However, we found no significant differences between the 12-cm and the 5-cm artificial pools in terms of species richness and community structure. It therefore appears that the design of drill-cored artificial rock pools may be determined by other factors (such as cost and time taken to drill shallow pools compared with deeper pools) without compromising their ecological function.

Conclusions and management implications

There is increasing recognition by policy makers of the need to incorporate ecologically sensitive design into marine and coastal developments (H.M. Government 2011; USACE 2012). In response, we trialled drill-cored artificial rock pools as a habitat enhancement intervention on an intertidal coastal defence breakwater. The desirability of different ecological responses to interventions will depend on specific secondary management objectives. Colonisation by species of conservation (e.g. *S. alveolata*) or commercial (e.g. *M. edulis*) value (as was observed in this study) may be positive for developments that seek to provide secondary socio-economic benefits or mitigate losses of natural habitats elsewhere. Similarly, colonisation by non-natives (e.g. *A. modestus*) (also observed in this study) may raise concern over the potential for interventions to facilitate the spread of invasive species (Airoldi and Bulleri 2011). In the absence of a single clear management objective from authorities (Moschella *et al.* 2005; Chapman and Underwood 2011; Firth *et al.* 2013a), we were interested in whether the artificial rock pools would increase biodiversity on the breakwater and support natural rock pool communities. We found that after 18 months of colonisation: (1) the artificial rock

pools supported greater species richness than adjacent rock surfaces on the breakwater; (2) the depth of artificial rock pools did not affect richness or community structure; and (3) the artificial rock pools supported equivalent species richness but different community compositions to natural rock pools on nearby rocky shores.

This habitat enhancement intervention provided important habitat for several taxa that were otherwise absent at mid-shore height on the breakwater, particularly mobile animals, lower-shore taxa and proportionally rarer taxa, that have all been noted as absent from coastal defences in previous studies (Chapman 2003; Moschella *et al.* 2005; Pister 2009). Although the artificial rock pools could not yet be considered fully functionally equivalent to natural rock pools, successional trajectories suggest that climax communities had not yet been reached – artificial and natural rock pool community structure may yet converge over time. In conclusion, these drill-cored artificial pools are an affordable, robust and effective means of enhancing biodiversity, and can be easily replicated in a variety of structures, both at the design stage and retrospectively.

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