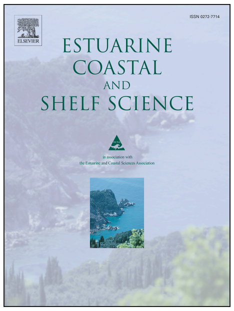

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# Ecological Enhancement Techniques to Improve Habitat Heterogeneity on Coastal Defence Structures

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## Abstract

Sea level rise and higher storm frequency are increasing the need for the placement of hard coastal defences worldwide. The majority of these defences lack optimal habitats for intertidal species, resulting in low diversity and abundance. The construction of coastal defences within marine protected areas (MPA) is also increasing and this study investigates ways to limit the loss of species diversity and intertidal habitat caused by installing rock armour defence structures and other coastal developments. Arrays of holes and grooves were created on granite rock armour in the north of England at Runswick Bay, N. Yorkshire and limestone rock groynes in southern England at Boscombe, Poole Bay, Dorset. Runswick Bay is a Marine Conservation Zone (MCZ) designated for its intertidal habitat and Boscombe is located in close proximity to a Special Area of Conservation (SAC). After 12 months, the treatments had attracted new species to the defence structures and increased the overall diversity and abundance of organisms compared to control areas. Mobile fauna including crabs and fish were also recorded utilising the holes and grooves at Boscombe. Non-native species were recorded in grooves at one site however their abundance was not significantly different to that of control areas. At the southern site, species known to be spreading in response to climate change were found in treatments but not in control areas. The cost of the installation of these enhancement techniques was low in relation to that of the defence scheme and could be easily incorporated before, during or after construction. Through evaluation of the use of these ecological enhancement techniques on coastal structures, it is suggested that they have considerable potential to increase biodiversity on artificial structures, particularly when used within large-scale coastal engineering defence projects.

*Keywords*: Ecological Enhancement, Structures, Biodiversity, Ecological Engineering, Marine Protected Area

# 1 Introduction

Sea level rise and higher storm frequency are increasing the need for hard coastal defences worldwide (Firth et al., 2016b). These structures are predominantly fabricated from materials that are novel to the local geology and marine environment and are designed to be durable and effective (French, 2001; Dong, 2004). Yet the construction of new defences may result in loss of intertidal habitat (Moschella et al., 2005). Hard coastal defence structures can form either a solid or permeable barrier, which can both absorb and dissipate wave energy, and are designed to provide a long-term cost- effective way of protecting land or assets from flooding and erosion (French, 2001). A variety of materials including concrete, wood and rock are used, although placement of rock armour boulders has more recently been favoured due to their longevity and efficiency at dispersing wave energy (Bradbury & Allsop, 1987; Crossman et al., 2003). The type of rock used in a particular area can be determined by the cost of transportation and aesthetic influences, particularly in marine protected areas (MPA). The design of coastal defence structures is informed by the specific erosion risks and local environmental conditions (Crossman et al., 2003; Garcia et al., 2004). In Europe, structures built within marine protected areas may be subject to formal Environmental Impact Assessment (85/337/EEC and 97/11/EEC), Habitat Directive Regulations (1992/43/EC) and the EU Water Framework Directive (2000/60/EC). Where there is a need to limit the loss of biodiversity caused by construction, coastal managers may be required to mitigate against any habitat loss.

Intertidal structures are typically colonised by sessile intertidal species, such as algae, barnacles, mussels and hydroids (Bacchiocchi & Airoldi, 2003; Bulleri & Chapman, 2004; Moschella et al., 2005; Mineur et al., 2012) with community composition differing due to the substrate type (Green et al., 2012), tidal height (Firth et al., 2013), wave exposure (Pister, 2009), orientation (Glasby & Connell, 2001) and location within a structure (Sherrard et al., 2016). The majority of structures lack surface heterogeneity and the ability to retain water at low tide (Bulleri & Chapman, 2004; Coombes et al., 2011; Firth et al., 2013, 2016b). In comparison, natural rocky shores generally have rougher surfaces and a variety of habitats including rock pools and crevices which provide refuge from both biotic and abiotic pressures at all states of tide (Raffaelli & Hawkins, 1996; Little et al., 2009; Firth et al., 2013; Aguilera et al., 2014). Barnacles are key space occupiers and habitat-forming species which occupy distinct zones on most UK intertidal rocky shores (Ballantine, 1961; Lewis, 1964) and the cold-temperate species *Semibalanus*. *Semibalanus balanoides* has been known to preferentially settle onto rough surfaces (Anderson & Underwood, 1994; Walters & Wethey, 1996; Holmes et al., 1997; Hills et al., 1999; Berntsson et al., 2000). The colonisation of these habitat-forming species then facilitate community succession and have positive impacts on species richness, abundance and community productivity (Jenkins et al., 1999; Thomsen et al., 2016). Limpets are key grazers on intertidal shores and control the abundance of algal species including ephemeral greens and fucoids (Raffaelli & Hawkins, 1996) Juvenile limpets are known to inhabit damp cracks and crevices until they reach 4-5 mm, at which point they move out onto drier rocks (Crump et al., 2003).

Adaptations can be made to coastal defence structures to encourage the colonisation and survival of intertidal species (Moschella et al., 2005; Dyson & Yocom, 2015), a process termed ‘ecological enhancement’ or ‘ecological engineering’ (Mitsch, 2012; Firth et al., 2014, 2016b; Sella & Perkol-Finkel, 2015; Strain et al., 2017). The purpose of ecological enhancement is to increase and/ or improve the habitat for biodiversity whilst also protecting human health and the environment (ITRC, 2004). Evans et al., (2017) found that ecological benefits were considered more important to stakeholders than socio-economic benefits when creating multifunctional structures. These adaptations can take many forms, including features that can be retrofitted on to existing structures (Firth et al., 2014, 2016b; Evans et al., 2015; Hall, 2017), perhaps within newly designated MPAs or be incorporated into the construction of new defence projects. In England, Marine Conservation Zones (MCZs) are created under The Marine and Coastal Access Act (2009) and if new structures were to be constructed within a MCZ ecological enhancement could be used to encourage colonisation of communities on the defence structure if appropriate.

Previous ecological engineering trials have aimed to improve the habitat heterogeneity of artificial structures through increasing the roughness of concrete (Coombes et al., 2015), drilling pits to seawalls (Martins et al., 2010, 2015), attaching precast concrete tiles (Borsje et al., 2011; Loke et al., 2015) in order to improve biodiversity (see Firth et al., 2016 for a review). Small scale water-retaining features have also been trialled by omitting blocks in the concrete (Chapman & Blockley, 2009), attaching flowerpots to seawalls (Browne & Chapman, 2011; Morris et al., 2017) core drilling pools in rock armour (Evans et al., 2015) and moulding concrete between boulders to form pools (Firth et al., 2016a). All of these interventions have had a measure of success in increasing the variety of habitats on the structures, resulting in either an increase in species richness or a change in community composition. On a larger scale, pre-cast habitat enhancement units have been trialled that incorporate rock pools of varying sizes, crevices and pits (Firth et al., 2014). Whilst these units can be incorporated into rock armour (Sella & Perkol-Finkel, 2015), it is difficult for them to be installed post-construction. This is important, as due to the prevalence of exsiting coastal defence structures, there is an outstanding need for low-cost retrofitting options, i.e. simple techniques which can be executed without large plant machinery or high construction costs, particularly in MPAs where disturbance from heavy machinery may damage features of the MPA. However, obtaining funding to retrofit improvements after the main project budget has been spent may be problematic, therefore, where possible, ecologcial enhancments should be incoroprated in the planning phase to enable adequate funds.

The current study evaluates the application of low-cost ecological enhancement techniques on coastal defence structures in sensitive marine habitats exposed to moderately high wave energy. In high wave energy environments, the use of rock armour (2-20 tonnes boulders) predominates and the attachment of artificial pools or tiles on the boulders is not an option as these could be removed by wave action, as already demonstrated in sheltered environments (Browne & Chapman, 2011). The low-cost treatments in this study are designed to be replicated on any boulder defence structure, including groynes, breakwaters and rock armour. These trials aimed to determine if these ecological enhancement techniques (“holes” and “grooves”) resulted in differences in community composition, species richness, total abundance, and species diversity of fauna and flora when compared to non-manipulated (Control) rock faces.

The following hypotheses were tested:

1) Species richness, total abundance and species diversity of fauna and flora would be greater in the treatment areas than prior to the treatment and in control areas. .

2) The community composition would vary between treatment and control areas.

3) There would be significantly more water retention in the treatment areas compared with the controls.

4) There would be an increased total abundance of habitat-forming functional groups (barnacles) and grazers (limpets) in the treatment areas compared to the controls.

# 2 Methods

## 2.1 Study Sites

Field trials were conducted to examine the ecological response of rocky shore species to two different enhancement treatments at each of two sites within the UK: Runswick Bay, North Yorkshire and Boscombe in Poole Bay, Dorset (Figure 1a). Runswick Bay was designated a Marine Conservation Zone (MCZ) (Marine and Coastal Access Act 2009) in 2016 for low energy intertidal rock, moderate energy intertidal rock, high energy intertidal rock and intertidal sand and muddy sand biotopes. Runswick Bay is a popular tourist area with a moderately exposed sandy shore and shale bedrock platforms approximately 100 m to the north of the test site. The existing rock granite armour consists of 5-10 tonne granite boulders sourced from the High Force Quarry in Middleton (UK), and was constructed in 2000 to dissipate wave energy and reduce overtopping of defences. Boscombe is located 11 km west of intertidal reef biotopes included within the Studland to Portland marine Special Area of Conservation (SAC) (EU Habitats Directive) designated in 2012. Boscombe has a moderately exposed urbanised coastline and is a popular tourist destination. It is predominantly sandy and the test site at Boscombe experiences a prevailing eastward longshore drift. The test site includes 3-6 tonne Portland limestone rock armour which was constructed in 2010 at Mean Low Water to strengthen the toe of older concrete groynes. Compared to nearby natural shores the rock armour at both study sites had a low abundance and diversity of colonising species (Authors personal observations), yet included barnacles and limpets that are important constituents of rocky shore ecosystems. Runswick Bay rock armour supported lower densities of barnacles, limpets and other intertidal molluscs compared to Boscombe, which had a more diverse community including barnacles, limpets, mussels and filamentous green, red and brown algae.

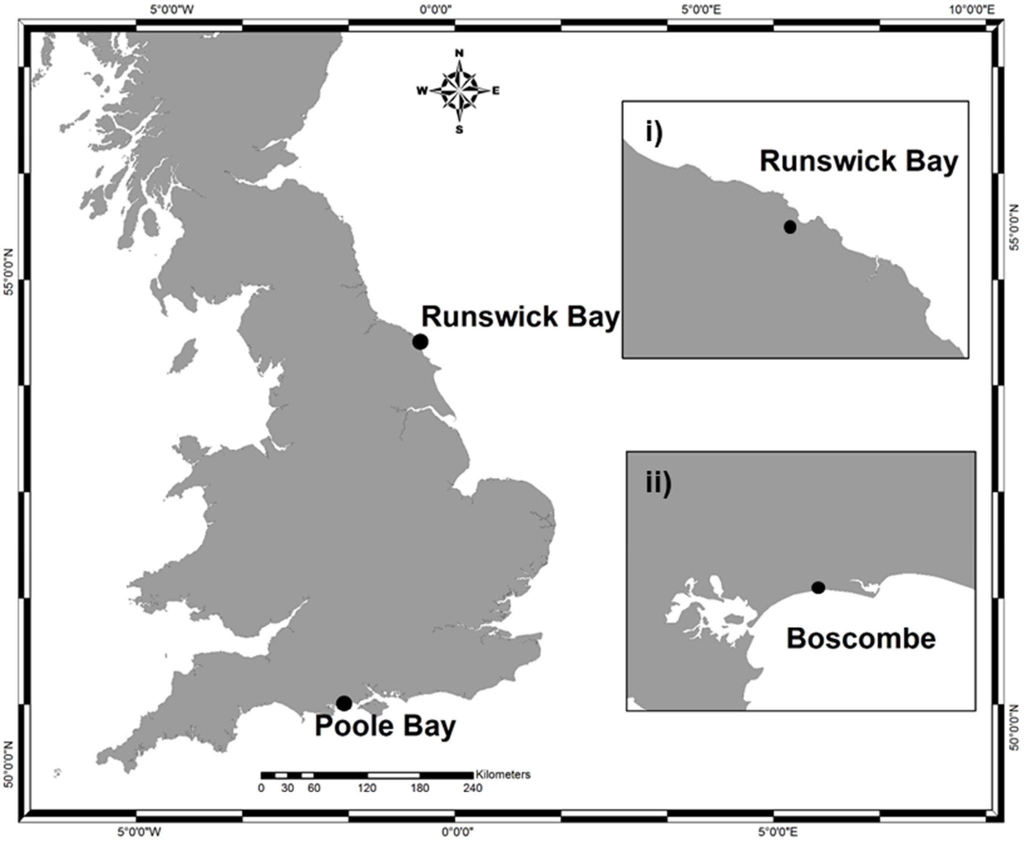


**Figure 1** Site locations of i) Runswick Bay and ii) Boscombe within Poole Bay, UK.

## 2.2 Interventions

Where logistically possible, treatments were created on the centre of the seaward surface of separate boulders. Two different enhancement treatments were evaluated at both sites.

(a) 'Holes', consisting of an array of four 20 mm deep x 16 mm diameter holes spaced 70mm apart, orientated to retain water at low tide, were drilled perpendicular into vertical surfaces of boulders using a rotary SDS hammer hand drill. Dimensions were chosen to mimic natural microhabitats observed on natural rocky shores.

(b) 'Grooves' aimed to replicate the groove-microhabitat occasionally observed in natural rocky shores and occasionally seen in rock armour as a consequence of use of explosives in the quarrying process. Each array consisted of two, thin horizontal grooves (approx. 60 cm long x 1 cm deep x 0.3 cm wide) and one thicker, coarser groove (approx. 60 cm long x 1 cm deep x 2 cm wide) that were cut in to the vertical surface of the rock using a petrol saw/angle grinder. The coarser middle grooves were chiselled out, which created a rough surface texture on the base and sides of the groove (Figure 1c). Both thin and thick grooves were included to provide a variety of habitats as observed in natural rocky shores.

(c) Control: At both sites, 20 x 20 cm control areas with similar orientation were created near each treatment on the same boulders by removing encrusting fauna and flora with a wire brush, paint-scraper and blow torch to create a bare surface.

## 2.3 Experimental Design

At Runswick Bay, two arrays of holes spaced 30 cm or more apart, were created on each of eight separate boulders (N=16) (Figure 1b, Figure 2a). In addition, three arrays of grooves were created on separate boulders (N=7) (Figure 1c, Figure 2b). All boulders were located between Mean Tide Level (MTL) and Mean Low Water at (MLW) and cleared of encrusting fauna and flora with a wire brush, paint-scraper and blow torch to create a bare surface prior to experimentation.

At Boscombe a larger trial was conducted in which two arrays of holes spaced 30 cm or more apart, were created on twenty-four boulders across two rock groynes which were situated 180 m apart (N=48) (Figure 1b, Figure 2c). In addition, three arrays of grooves were created on twenty-four separate boulders located across two groynes (N=24) (Figure 1c, Figure 2d). All boulders were located at Mean Low Water at (MLW) and cleared of encrusting fauna and flora with a wire brush, paint-scraper and blow torch to create a bare surface prior to experimentation.

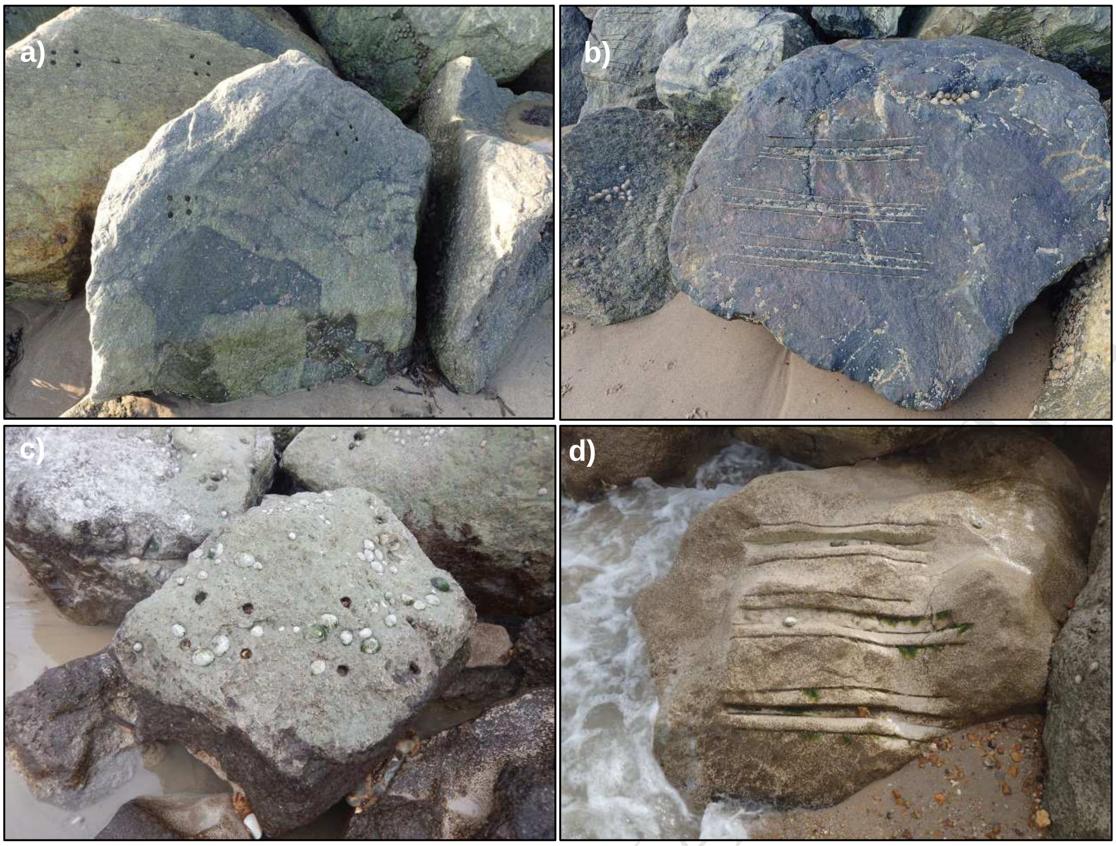


**Figure 2** Positioning of a) Holes at Runswick Bay, b) Grooves at Runswick Bay, c) Holes at Boscombe, d) Grooves at Boscombe.

The cost of the treatments in Boscombe was £500 (€570, $700 USD), which covered two workers' for 4 hours, tool hire and a replacement blade/ drill bit. At Runswick Bay the structures were built of granite so the time taken to complete the enhancements was longer than at Boscombe due to the hardness of the rock, so less replication of treatments was undertaken. In addition, diamond tipped drill bits and blades were needed to create the treatments which were included in the overall cost of £660 (€750, $924 USD).

## 2.4 Surveillance

At both sites, boulders on each structure were thoroughly surveyed using 20 x 20cm quadrats to record the percentage cover of seaweed and counts of fauna prior to the installation of treatments. Treatments and controls were established in October 2014 at

Runswick Bay and March 2015 at Boscombe, and then sampled after one year. The boulders with holes were sampled using a 20 x 20 cm quadrat placed over each array and control areas and the percentage cover of seaweed and counts of fauna, such as barnacles, limpets, mussels and smaller gastropods were recorded to measure species abundance.

For boulders with grooves, nine 20 x 20 cm quadrats were placed on the treatment area and on the adjacent control areas and the percentage cover of seaweed and counts of fauna were recorded, from which a mean abundance was calculated for both treatment and control. Percentage cover of water retention and sediment in each treatment and control quadrat was also recorded through visual estimates. During each survey, a record of all species observed on the whole of each structure at both sites was made to determine whether any new species colonised the structures as a result of the treatments.

An estimate of surface heterogeneity of the rocks (in order to account for the increased surface area due to treatments) in each sampled quadrat was made at the start of the experiment using a fine scale variation of the chain and transect method (Luckhurst & Luckhurst, 1978; Frost et al., 2005). A thin chain was secured at the top of the quadrat and run to the bottom edge ensuring it touched the bedrock. This distance was then measured and used as a measure of relative surface texture (space available for colonisation (Loke & Todd, 2016)) within each quadrat sampled.

## 2.5 Statistical Analysis

To account for the increased surface area provided through the installation of holes and grooves onto a boulder surface, a correction factor was applied to standardise all abundance data of flora and fauna collected from treatment quadrats. This was calculated using an average of the surface area measurements collected across all quadrats for each treatment. The correction factor applied to abundance data was 0.8 for quadrats containing grooves and 0.82 for quadrats containing holes.

Species richness, total abundance of fauna and flora and Shannon-Weiner species diversity indices were determined using the DIVERSE function in PRIMER-e V6 (Clarke, 2001). A one-way ANOVA was performed for each treatment and site separately with treatment (Before vs Holes/Grooves vs Control) as the main factor (Long & Ervin, 2000). Any significant effects were explored using a Tukey post hoc test. A Bray Curtis similarity matrix was generated from square-root transformed data and the ANOSIM procedure used to test if there was any significant difference in communities of benthic organisms between treatments (Clarke, 2001). The SIMPER routine was performed for each site separately to determine species contributing most to the similarity within treatments and dissimilarity between treatments and controls (Clarke, 2001).

To determine if there was a difference in the average number of barnacles and limpets recorded in the different treatments versus the control areas, a negative binomial Generalised Liner Model (GLM) was applied for each site separately. Due to numerous zero observations in count data the application of the negative binomial model resolved issues relating to over-dispersion and had the lowest Akaike Information Criterion (AIC) of the models trialled and, after examination of the residuals, was determined to be the most applicable to the data (Zuur et al., 2009). All analyses were undertaken in R Studio using the MASS routine (Venables & Ripley, 2002) and base package (R Core Team, 2016).

# 3 Results

## 3.1 Runswick Bay – granite rock armour

Only 2 species were recorded on the boulders before the treatments were installed (Table 1), yet following the treatments an additional 6 species were observed to have colonised the holes and an additional 5 species in the grooves. These new species included algae *Porphyra* sp., *Fucus* sp. and *Mastocarpus stellatus*, two gastropod snail species *Littorina* *saxatilis* and *Melarhaphe neritoides* and the mussel *Mytilus edulis* (Table 1), all but *Fucus* sp. and *Mastocarpus stellatus* were also found in the control areas.

**Table 1** Presence and absence of species after a 12 month period for before, holes, grooves and controls at Runswick Bay and Boscombe (* indicates presence after 12months, + indicates presence between 0-12months).

| **Group** | **Species** | **Runswick Bay** | | | | **Boscombe** | | | |
|---|---|---|---|---|---|---|---|---|---|
| | | Before | Holes | Grooves | Control | Before | Holes | Grooves | Control |
| Algae | *Ceramium* sp. | | | | | | * | * | * |
| | *Chaetomorpha* sp. | | | | | | | * | |
| | *Cladophora rupestris* | | | | | | | * | |
| | *Codium fragile* | | | | | | | + | + |
| | Diatom | | | | | * | + | * | * |
| | *Dumontia cortorta* | | | | | | | + | |
| | *Fucus* sp. | | * | + | | | | | |
| | *Halurus* sp | | | | | | | + | + |
| | *Lomentaria articulata* | | | | | | | + | + |
| | *Mastocarpus stellatus* | | | + | | | | | |
| | *Polysiphonia* sp. | | | | | | + | * | |
| | *Porphyra* sp | | * | * | * | | + | * | |
| | *Rhodochorton purpureum* | | | + | * | | | | |
| | *Rhodothamniella floridula* | | | + | | | + | * | * |
| | *Scytosiphon lomentaria* | | | | | | + | + | + |
| | *Ulva lactuca* | | | + | | * | | * | + |
| | *Ulva linza* | | * | * | * | * | * | * | * |
| Cnidaria | *Actina equina* | | | | | | | * | + |
| | *Anemonia viridis* | | | | | | | * | |
| Annelida | *Eulalia viridis* | | | | | | | * | |
| | *Polydora ciliata* | | | | | | + | * | |
| | *Spirobranchus triqueter* | | | | | | * | * | * |
| Crustacean | *Austrominius modestus* | | | | | | | * | * |
| | *Perforatus perforatus* | | | | | | + | * | |
| | *Carcinus maenas* | | | | | | * | + | |
| | *Idotea granulosa* | | | + | | | | | |
| | *Semibalanus balanoides* | * | * | * | * | * | * | * | * |
| Mollusca | *Lepidochitona cinereus* | | | | | | + | * | |
| | *Littorina saxatilis* | | * | * | * | | | | |
| | *Melarhaphe neritoides* | | * | * | * | | | | |
| | *Mytilus edulis* | | * | * | * | * | * | * | * |
| | *Nucella lapillus* | | | | | | * | + | |
| | *Patella depressa* | | | | | | | | * |
| | *Patella vulgata* | * | * | * | * | * | * | * | * |
| | *Rissoa* sp. | | | | | | | + | + |
| Bryozoa | Bryozoa sp. | | | | | | * | * | |
| Ascidicea | *Ascidiella aspersa* | | | | | | * | + | |
| Chordata | *Lipophrys pholis* | | | | | | * | | |
| **Total Number of Species observed between 0-12months** | | **2** | **8** | **13** | **8** | **6** | **19** | **30** | **17** |
| **Total Number of Species after 12 months** | | **2** | **8** | **7** | **8** | **6** | **11** | **21** | **10** |

There was a significantly greater species richness, Shannon-Weiner species diversity and total abundance of fauna and flora in the holes (Table 2a; Figure 3a) compared to before (P<0.001) and the controls (P<0.001). The grooves treatments supported a greater species richness and total abundance of organisms when compared to before and the controls (P<0.001), alongside supporting a higher Shannon-Weiner species diversity than before, however there were no significant difference in Shannon-Weiner species diversity between grooves and controls (Table 2a; Figure 3a). Both treatments created novel areas of water retention which were lacking on the control sites (Figure 4).

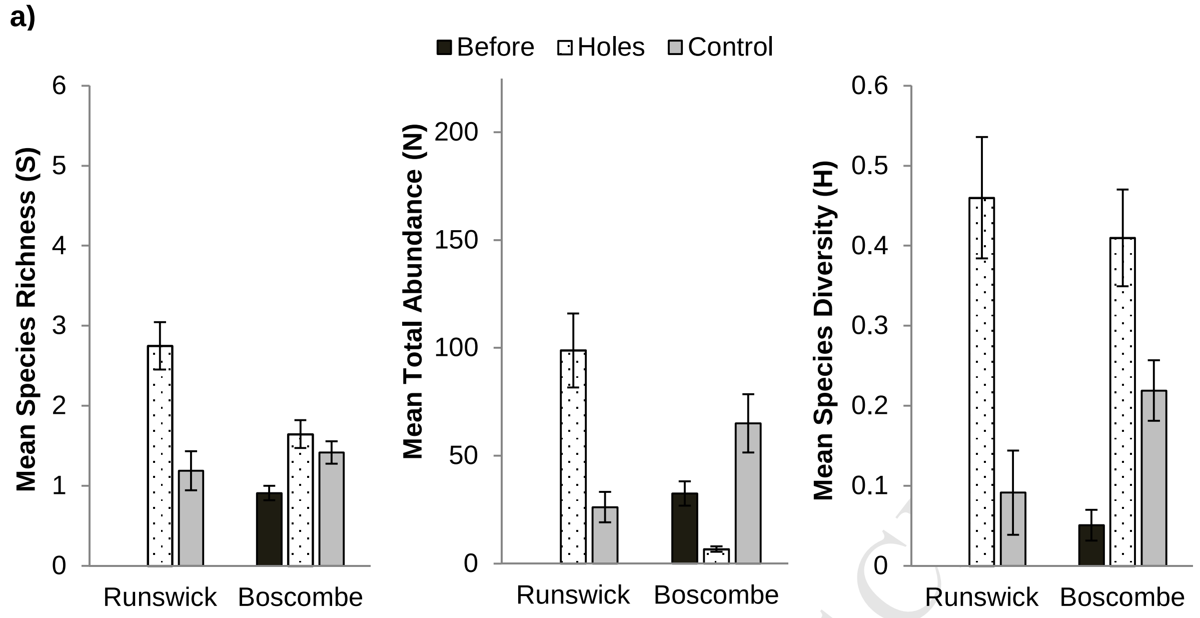
**Table 2** Results of one way ANOVA for comparison in species richness, total abundance and species diversity (H) in before, holes and control quadrats and before, grooves and control quadrats at a) Runswick Bay and b) Boscombe after 12 months.

**a) Runswick Bay**

| | **Species richness** | | | **Total abundance** | | | **Species diversity** | | |
|---|---|---|---|---|---|---|---|---|---|
| | *df* | F | p | *df* | F | p | *df* | F | p |
| Holes | *2* | 38.65 | <0.001 | *2* | 22.80 | <0.001 | *2* | 20.91 | <0.001 |
| **Contrasts** | | | | | | | | | |
| Before- Holes | *45* | | <0.001 | *45* | | <0.001 | *45* | | <0.001 |
| Holes - Control | 45 | | <0.001 | *45* | | 0.001 | *45* | | <0.001 |
| | | | | | | | | | |
| Grooves | *2* | 165.8 | <0.001 | *2* | 33.61 | <0.001 | *2* | 3.48 | 0.052 |
| **Contrasts** | | | | | | | | | |
| Before- Grooves | *18* | | <0.001 | *18* | | <0.001 | *18* | | 0.046 |
| Grooves -Control | *18* | | <0.001 | *18* | | <0.001 | *18* | | 0.670 |

**b) Boscombe**

| | **Species richness** | | | **Total abundance** | | | **Species diversity** | | |
|---|---|---|---|---|---|---|---|---|---|
| | *df* | F | p | *df* | F | p | *df* | F | p |
| Holes | *2* | 7.80 | <0.001 | *2* | 12.07 | <0.001 | *2* | 16.91 | <0.001 |
| **Contrasts** | | | | | | | | | |
| Before- Holes | *141* | | <0.001 | *141* | | 0.253 | *141* | | <0.001 |
| Holes - Control | *141* | | 0.489 | *141* | | <0.001 | *141* | | 0.006 |
| | | | | | | | | | |
| Grooves | *2* | 27.86 | <0.001 | *2* | 0.23 | 0.794 | *2* | 6.91 | 0.001 |
| **Contrasts** | | | | | | | | | |
| Before- Grooves | *69* | | <0.001 | *69* | | 0.88 | *69* | | 0.001 |
| Grooves -Control | *69* | | <0.001 | *69* | | 0.78 | *69* | | 0.08 |



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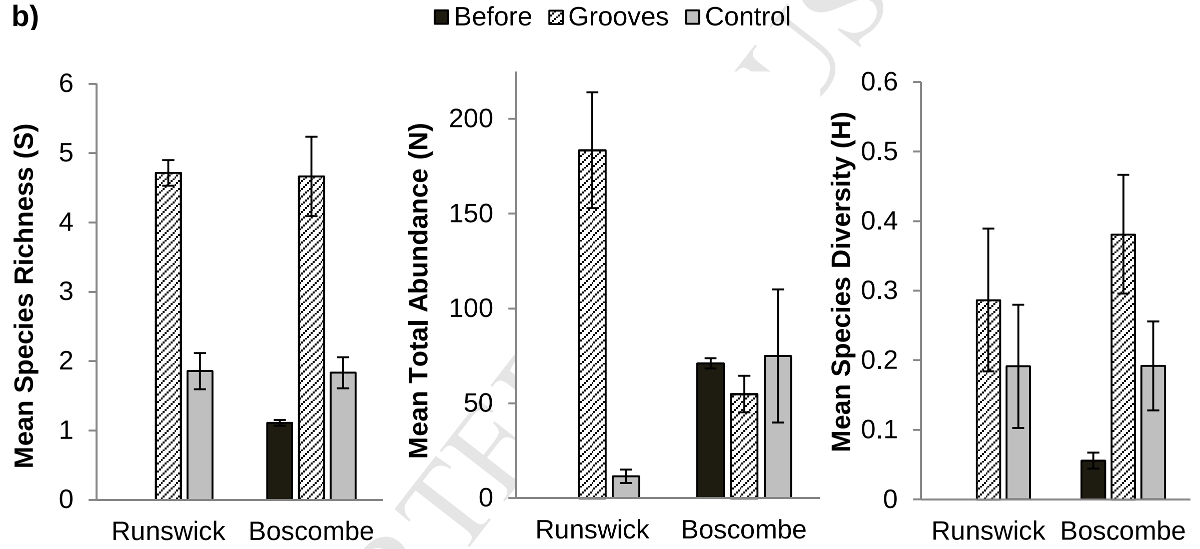

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286 **Figure 3** Mean species richness (S), total abundance (N) and species diversity (H) for a) holes and b) grooves before installation compared to the test and control after 12 months at Runswick Bay and Poole Bay (+/- SE).

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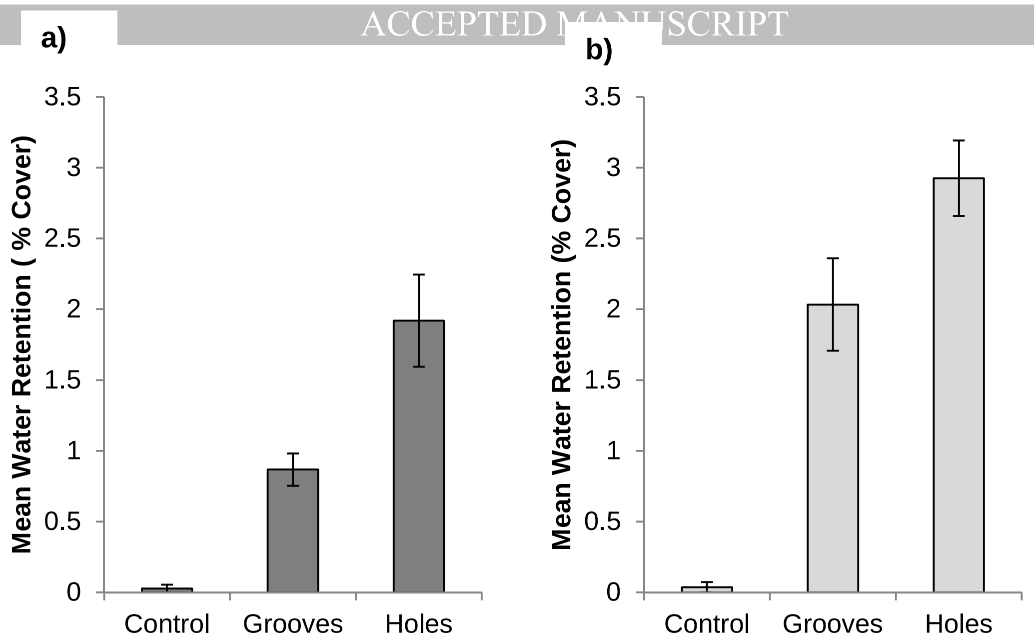

**Figure 4** Mean percentage of water retention for the control, holes and grooves at a) Runswick Bay and b) Boscombe (Mean +/- S.E.).

Community similarity was found to be significantly different between the holes and controls (ANOSIM, R=0.17, P<0.01) and grooves and controls (ANOSIM, R=0.95, P<0.02) after 12 months. Of the overall 84.5% dissimilarity between holes and control, 98.8% could be attributed to the higher abundance *of Semibalanus balanoides, Ulva linza, Melarhaphe neritoides, Littorina saxatilis* and *Mytilus edulis* in the holes (Table 3a). Whereas 98.9% of the overall 86.6% dissimilarity between grooves and control was attributed to greater abundance of *S. balanoides, Ulva linza* and *Melarhaphe neritoides* in the grooves (Table 3b). There were significantly higher counts of habitat-forming barnacles in both the grooves and holes treatments compared to the controls (Table 4a & Figure 5). No significant difference was found for limpet abundance (Table 4b & Figure 5b).

**Table 3** SIMPER table indicating average abundance of species per array in a) Holes and Control b) Grooves and Control at Runswick Bay after 12 months (Av.Abund= Mean Abudance (Raw), Av.Diss = Average Dissimiliarity, Diss/SD= Dissimilarity SD, Contrib%=Contribution percentage, Cum %= Cumulative percentage, c= counts, & = percentage cover).

| **a) Holes & Control** | | | | | **Average dissimilarity = 84.49** | |
|---|---|---|---|---|---|---|
| **Species** | **Holes Av.Abund** | **Control Av.Abund** | **Av.Diss** | **Diss/SD** | **Contrib%** | **Cum.%** |
| *Semibalanus balanoides* (c ) | 76.44 | 12.18 | 56.27 | 1.8 | 66.6 | 66.6 |
| *Ulva linza* (%) | 5.67 | 13.64 | 13.74 | 0.55 | 16.26 | 82.87 |
| *Melarhaphe neritoides ( c )* | 5.49 | 0.82 | 5.91 | 0.63 | 6.99 | 89.85 |
| *Littorina saxatilis* (c ) | 5.67 | 0.09 | 5.37 | 0.73 | 6.36 | 96.21 |
| *Mytilus edulis* (c ) | 2.95 | 0.05 | 2.2 | 0.41 | 2.61 | 98.82 |
| *Porphyra* sp. (%) | 0.22 | 1.05 | 0.83 | 0.5 | 0.98 | 99.8 |
| *Fucus* sp. (%) | 0.07 | 0.00 | 0.08 | 0.21 | 0.1 | 99.9 |
| *Patella vulgata* (c ) | 0.07 | 0.00 | 0.08 | 0.3 | 0.1 | 100 |
| **b) Grooves & Controls** | | | | | **Average dissimilarity = 86.61** | |
| **Species** | **Grooves Av.Abund** | **Control Av.Abund** | **Av.Diss** | **Diss/SD** | **Contrib%** | **Cum.%** |
| *Semibalanus balanoides* (c ) | 174.22 | 11.06 | 79.01 | 4.30 | 91.23 | 91.23 |
| *Ulva linza* (%) | 3.67 | 0.00 | 3.97 | 0.48 | 4.59 | 95.82 |
| *Melarhaphe neritoides ( c )* | 4.22 | 0.00 | 2.63 | 1.32 | 3.03 | 98.85 |
| *Littorina saxatilis* (c ) | 1.31 | 0.24 | 0.85 | 0.83 | 0.98 | 99.83 |
| *Patella vulgata* (c ) | 0.08 | 0.11 | 0.09 | 0.69 | 0.10 | 99.93 |
| *Rhodochorton purpureum (%)* | 0.00 | 0.08 | 0.04 | 0.36 | 0.05 | 99.98 |
| *Mytilus edulis* (c ) | 0.03 | 0.00 | 0.01 | 0.59 | 0.01 | 99.99 |
| *Porphyra* sp. (%) | 0.01 | 0.00 | 0.00 | 0.40 | 0.01 | 100.00 |

**Table 4** Summary of the results of the negative binomial GLM applied to i) barnacle and ii) limpet count data with treatment as the factor at a) Runswick Bay b) Boscombe (***= P<0.001, **= P<0.01, NS=Not significant).

**a) Runswick Bay**

| **i) Barnacles** | | | **AIC=517.33, Theta=0.472** | |
|---|---|---|---|---|
| | **Estimate** | **Std. Error** | **Z value** | **P value** |
| Intercept | 2.477 | 0.275 | 8.998 | *** |
| Grooves | 2.682 | 0.615 | 4.359 | *** |
| Holes | 1.859 | 0.145 | 4.475 | *** |
| **ii) Limpets** | | | **AIC=29.412, Theta=1962** | |
| Intercept | -3.615 | 1.323 | -3.193 | ** |
| Grooves | 1.053 | 1.770 | 0.595 | NS |
| Holes | 0.994 | 1.381 | 0.720 | NS |

**b) Boscombe**

| **i) Barnacles** | | | **AIC= 465.95 Theta= 0.059** | |
|---|---|---|---|---|
| | **Estimate** | **Std. Error** | **Z value** | **P value** |
| Intercept | 2.782 | 0.483 | 5.751 | **** |
| Grooves | -0.034 | 0.967 | -0.036 | NS |
| Holes | -2.276 | 0.772 | -2.942 | ** |
| **ii) Limpets** | | | **AIC=476.92, Theta=0.282** | |
| Intercept | 0.214 | 0.246 | 0.872 | NS |
| Grooves | 0.257 | 0.484 | 0.532 | NS |
| Holes | 0.850 | 0.376 | 2.259 | * |

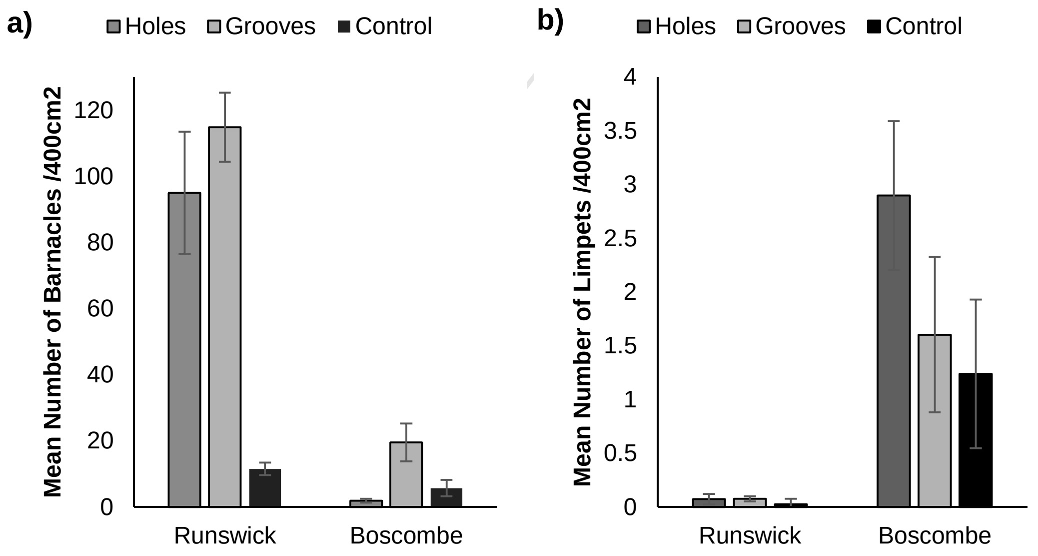

**Figure 5** Mean abundance of a) barnacles and b) limpets in the holes, grooves and control quadrats at Runswick Bay and Boscombe (Count data, Mean +/- S.E).

## 3.2 Boscombe, Poole Bay– limestone rock armour

The rock groyne boulders at Boscombe supported 6 taxa before the treatments were installed and, after 12 months, 11 taxa were recorded in the holes and 21 taxa in the grooves and 10 taxa recorded in the control areas (Table 1). Species that were only found within the holes and groove treatments and observed nowhere else on the structures included *Ascidiella aspersa, Anemonia viridis, Carcinus maenas* and a bryozoan (Table 1). Overall, there was a significant difference in species richness and species diversity before and after the holes treatment (Table 2), yet there was no difference in total abundance. There was however a significant difference in total abundance and species diversity between the holes and control quadrats after 12 months (Table 2). The groove treatments showed a significant difference in species richness and species diversity in quadrats before and after the treatment and a significant difference in species richness between the treatment and control areas (Table 2). The grooves treatment at Boscombe resulted in the greatest increase in species diversity compared to that present prior to the treatment and the control quadrats. The non-native barnacle species *Austrominius modestus* was only recorded in the control and grooves quadrats.

Community similarity was found to be significantly different between the holes and controls (ANOSIM, R=0.07, P<0.02) but not between the grooves and controls (ANOSIM, R=0.01, P>0.05) after 12 months. Four species accounted for 87.8% of the overall 91% dissimilarity between holes and controls, there was a greater abundance of *Ulva linza, Semibalanus balanoides* and *Rhodthamniella fluoridula* in the control areas and a higher number of *Patella vulgata* in the holes treatment (Table 5). Six taxa were only recorded in the holes and not the control areas, these were the crab *Carcinus maenas,* sea squirt *Ascidiella aspersa,* gastropod *Nucella lapillus,* bivalve *Mytilus edulius*, Bryozoan and the fish *Lipophrys pholis.*

The variation in communities between the grooves and control areas was attributed to 22 taxa (Table 5). Of the overall 80% dissimilarity between grooves and control, 90.5% could be attributed to the greater abundance of Diatom and *Rhodothamniella floridula* in the controls and a higher abundance of *Semibalanus balanoides* in the grooves (Table 5). The grooves supported 14 taxa which were absent from the controls, these included the chiton *Lepidochitona cinereus,* the anemone *Actina equina* and the barnacle *Perforatus perforatus.* There were significantly lower numbers of barnacles found in the holes quadrats compared to the control (Table 4b, Figure 5). However, the number of limpets was significantly higher in the holes treatment compared to the control and grooves samples (Table 4b).

**Table 5** SIMPER table indicating average abundance of species per array in a) Holes and Control b) Grooves and Control at Boscombe, Poole Bay after 12 months (Av.Abund= Mean Abudance (Raw), Av.Diss = Average Dissimilarity, Diss/SD= Dissimilarity SD, Contrib%=Contribution percentage, Cum %= Cumulative percentage, c= counts, & = percentage cover).

| **a) Holes & Control** | | | | | | **Average dissimilarity = 91.01** |
|---|---|---|---|---|---|---|
| **Species** | **Holes Av.Abund** | **Control Av.Abund** | **Av.Diss** | **Diss/SD** | **Contrib%** | **Cum.%** |
| *Ulva linza* (%) | 0.52 | 25.42 | 32.26 | 0.86 | 34.09 | 34.09 |
| *Semibalanus balanoides* (c ) | 1.67 | 23.33 | 20.76 | 0.63 | 21.94 | 56.03 |
| *Patella vulgata* (c ) | 2.90 | 1.29 | 18.70 | 0.64 | 19.76 | 75.79 |
| *Rhodothamniella floridula* (%) | 0.00 | 12.92 | 11.32 | 0.45 | 11.96 | 87.75 |
| *Spirobranchus triqueter* (c ) | 1.15 | 0.08 | 4.87 | 0.37 | 5.14 | 92.90 |
| *Ceramium* sp. (%) | 0.20 | 1.96 | 3.57 | 0.40 | 3.77 | 96.67 |
| *Carcinus maenas* (c ) | 0.05 | 0.00 | 1.10 | 0.14 | 1.16 | 97.83 |
| *Ascidiella aspersa* (c ) | 0.08 | 0.00 | 0.58 | 0.15 | 0.61 | 98.44 |
| *Nucella lapillus* (c ) | 0.02 | 0.00 | 0.56 | 0.08 | 0.59 | 99.03 |
| *Mytilus edulius* (c ) | 0.07 | 0.00 | 0.39 | 0.16 | 0.42 | 99.44 |
| Bryozoan (%) | 0.05 | 0.00 | 0.35 | 0.10 | 0.37 | 99.81 |
| *Lipophrys pholis (c )* | 0.02 | 0.00 | 0.18 | 0.10 | 0.19 | 100.00 |
| **b) Grooves & Controls** | | | | | | **Average dissimilarity = 80.03** |
| **Species** | **Grooves Av.Abund** | **Control Av.Abund** | **Av.Diss** | **Diss/SD** | **Contrib%** | **Cum.%** |
| Diatom (%) | 23.77 | 58.44 | 38.21 | 1.13 | 47.75 | 47.75 |
| *Rhodothamniella floridula* (%) | 11.20 | 13.52 | 22.10 | 0.73 | 27.61 | 75.36 |
| *Semibalanus balanoides* (c ) | 15.61 | 1.81 | 12.14 | 0.58 | 15.17 | 90.53 |
| *Patella vulgata* (c ) | 1.60 | 1.13 | 4.01 | 0.39 | 5.01 | 95.54 |
| *Spirobranchus triqueter* (c ) | 1.96 | 0.01 | 2.62 | 0.45 | 3.27 | 98.81 |
| *Austrominius modestus* (c ) | 0.14 | 0.06 | 0.24 | 0.43 | 0.30 | 99.10 |
| *Ulva linza* (%) | 0.21 | 0.00 | 0.21 | 0.54 | 0.26 | 99.37 |
| *Ceramium* sp. (%) | 0.17 | 0.00 | 0.18 | 0.44 | 0.22 | 99.59 |
| *Mytilus edulis* (c ) | 0.10 | 0.01 | 0.15 | 0.49 | 0.19 | 99.78 |
| *Ulva lactuca* (%) | 0.04 | 0.00 | 0.04 | 0.38 | 0.04 | 99.82 |
| *Lepidochitona cinereus* (c ) | 0.02 | 0.00 | 0.02 | 0.38 | 0.03 | 99.86 |
| *Polysiphonia* sp. (%) | 0.02 | 0.00 | 0.02 | 0.19 | 0.03 | 99.88 |
| *Actina equina* (c ) | 0.01 | 0.00 | 0.02 | 0.18 | 0.02 | 99.91 |
| Bryozoan (%) | 0.02 | 0.00 | 0.01 | 0.20 | 0.02 | 99.92 |
| *Porphyra* sp. (%) | 0.01 | 0.00 | 0.01 | 0.18 | 0.02 | 99.94 |
| *Patella depressa* (c ) | 0.00 | 0.01 | 0.01 | 0.20 | 0.01 | 99.95 |
| *Cladophora rupestris* (%) | 0.01 | 0.00 | 0.01 | 0.26 | 0.01 | 99.97 |
| *Eulalia viridis* (c ) | 0.00 | 0.00 | 0.01 | 0.18 | 0.01 | 99.98 |
| *Pseudopolydora pulchra* (c ) | 0.00 | 0.00 | 0.01 | 0.18 | 0.01 | 99.98 |
| *Perforatus perforatus* (c ) | 0.01 | 0.00 | 0.01 | 0.28 | 0.01 | 99.99 |
| *Anemonia viridis* (c ) | 0.00 | 0.00 | 0.00 | 0.20 | 0.00 | 100.00 |
| *Chaetomorpha* sp. (%) | 0.00 | 0.00 | 0.00 | 0.20 | 0.00 | 100.00 |

# 4 Discussion

The holes and grooves ecological enhancement techniques on both the granite rock armour at Runswick Bay and the limestone rock groynes at Boscombe supported significantly greater species richness and diversity compared to the un-manipulated control areas (Tables 2 and 4). The creation of holes on the boulders significantly increased total abundance of organisms on both artificial structures (Tables 2 and 4), whereas total abundance in the grooves treatment was only significantly different for the granite boulders at Runswick Bay. The type of rock used to construct coastal defence structures has been shown to affect community composition, with hard, fine-grained rocks, such as granite and basalts, supporting less diverse communities than sandstones (Green et al., 2012) and limestones (Sherrard et al., 2016). Yet, the greater species richness observed on the limestone boulders in the English Channel at Boscombe compared to the granite boulders at Runswick Bay in the North Sea can also be attributed to biogeographical differences and sea temperature (Forbes & Goodwin-Austen, 1859; Southward et al., 1995; Herbert et al., 2003; Hawkins et al., 2009). Softer rocks, such as limestone, naturally weather to create crevices and rough surfaces, whereas harder rock, such as granite, weather more slowly, leaving smooth, flat rock faces that are less favourable to species settlement and colonisation (Berntsson et al., 2000; Moschella et al., 2005; Herbert & Hawkins, 2006). The quarrying process of cutting rock to size also produces smooth surfaces with little surface heterogeneity and so until significant weathering occurs, surface roughness will remain low (Coombes et al., 2011, 2015), resulting in variation of communities with age of the structures (Moschella et al., 2005; Pinn et al., 2005). The increased heterogeneity resulting from the treatments on the granite boulders at Runswick Bay enhanced colonisation resulting in a marked increase in richness, abundance and diversity. Although variation in species richness has previously been observed on the inside and outside faces of limestone boulders used for rock groynes (Sherrard et al., 2016), this was not assessed in this study.

Whilst a significant increase in number of barnacles occupying the grooves was observed on both shores (Table 3), this was not the case for the holes treatment. Barnacle settlement has been shown to be greater on rough surfaces, whilst mobile intertidal snails (e.g. *Littorina saxatilis*) actively select a groove or hole in a rock compared to a bare rock surface with no refuge (Pardo & Johnson, 2004; Martins et al., 2010; Skov et al., 2011). In the current trials, newly settled and mobile species were found in greater abundance in the treatment areas compared with the bare rock faces and the before communities. The treatments used in the current trial not only introduced additional substrate heterogeneity and rugosity, but also created areas of water retention (Figure 4). The lack of water retention and available refuges on artificial structures has previously been shown to result in reduced species richness (Bulleri & Chapman, 2004; Coombes et al., 2011; Firth et al., 2013; Aguilera et al., 2014). On a granite breakwater, Evans *et al.,* (2015), revealed that artificial pools supported equivalent species richness to the nearby natural rock pools and were shown to create suitable habitat for species previously absent from the artificial structure at mid-shore height. The results here support this, as new species were also recorded in the holes and grooves at both sites that were previously absent from the boulders. Firth et al., (2013a) found that rock pools in artificial structures have a more pronounced effect on species richness in both the mid and upper-shore zones. This suggests that modifications will have the greatest impact in the upper and mid shore habitats.

Limpets, however, did not show an increase in abundance with all treatments, which was attributed to the small amount of space in the holes, resulting in a limited size and abundance of individuals able to utilise them (See Methods section 2.2 for dimensions). At Boscombe, the number of limpets was significantly higher in areas which included the holes treatment, but the same effect was not observed at Runswick Bay.  Furthermore, the grooves at Boscombe regularly trapped stones, shells and sand which could both encourage and deter species from colonising (Airoldi & Hawkins, 2007; Liversage et al., 2017). The additional refuge created by shell and stone debris could facilitate development of algal propagules (Bulleri, 2005) and colonisation by small gastropod snails, yet prevent refuge for large species such as limpets and fish. Overall, the use of these simple treatments had a positive effect on richness and diversity of marine life on rock armour structures and enhanced the colonisation of common rocky shore species.

The reduced abundance of mobile fauna has previously been noted on artificial structures which results from low habitat heterogeneity and limited refugia (Chapman, 2003). Here, the addition of holes and grooves resulted in previously absent mobile fauna to be recorded on the groynes, including fish (*Lipophrys pholis*) and crabs (*Carcinus maenas*) in the holes of the Boscombe treatment. At Boscombe, the limpets (especially juveniles typically less than 16 mm) favoured the holes that acted as refugia until they had outgrown the hole, when they could potentially migrate onto the surrounding rock surface. In the Azores, Martins et al., (2010) showed that holes can be used to successfully attract and harvest limpets for human consumption. Several algal species, including *Fucus* sp. and *Mastocarpus stellatus,* that attached to the rough textures within the grooves, were absent on the bare rock faces. The creation of rough surfaces as a consequence of these interventions allowed algal propagules to attach and 'escape' due to the refuge provided from predators, dislodgement and desiccation (Hawkins, 1981; Moore et al., 2007). The presence of macrophytes such as *Fucus* spp. will encourage subsequent mobile fauna, as the alga provides refuge from predators and desiccation (Christie et al., 2009).

The community establishment of an artificial structure will be dependent on season as larval and propagule supply will effect subsequent community development (Moschella et al., 2005; Pinn et al., 2005). The timing of ecological enhancements needs to be considered as this will determine the community establishment and development. As coastal defence structures are commonly constructed in high wave energy environments, the communities formed on hard structures can be stripped back to a bare substratum during storm events or maintenance activities (Sousa, 1979). The development and survival of these communities will depend on the impacts particular species may have on community development prior to their arrival (priority effects) which are determined by biological and environmental conditions (Hall, 2015). Consequent changes in communities could be observed in subsequent months and years due to succession and disturbances, reinforcing the need for long term monitoring (Sheehan et al., 2013).

It has been established that artificial structures support less diverse communities than natural rocky shores (Chapman & Bulleri, 2003; Bulleri & Chapman, 2004; Moschella et al., 2005; Glasby et al., 2007; Vaselli et al., 2008; Firth et al., 2013). Following an initial colonisation of microbial film, structures are colonised by larger opportunistic species such as *Ulva* spp. with subsequent community development then dependent on local conditions and propagule supply (Benedetti-Cecchi, 2000). In the current study, the holes and grooves trials resulted in an increase in richness and diversity, irrespective of geology (See Table 2 and Figure 3), indicating that even simple measures can have a beneficial effect on the biodiversity of a rock armour structure. The nature of the enhancement technique also means that this can be implemented at any stage during the life history of the coastal defences, adding biodiversity to existing structures as well as being incorporated into new ones.

There has been concern that artificial structures can increase the spread and abundance of non-native species (Bulleri & Airoldi, 2005) which could be detrimental, however in the current study the number of non-native species recorded at both sites was low. Non-native species were not recorded at Runswick Bay, either in previous baseline surveys, treatments or controls. The barnacle *Austrominius modestus* was found in both the holes and grooves treatments in Boscombe but in numbers comparable to control areas and densities across the structures. The increased interspecific competitive and predatory interactions resulting from higher species diversity associated with these treatments may limit populations of invasive species on these structures (Levine, 2000); however this was not confirmed at the scale of these experiments.

Climate migrants, such as *Gibbula umbilicalis* whose range is expanding in response to rising temperatures (Keith et al., 2011) may benefit from such treatments (Hawkins et al., 2009). Both the warm-temperate barnacle species *Perforatus perforatus* and sea anemone *Anemonia viridis* were found in some of the treatments at Boscombe, but could not be found elsewhere on the groyne rock armour. The increased surface texture created by the treatments could facilitate further expansion of climate migrants as they provide refugia (Bourget et al., 1994) and could promote establishment. The increased effects of climate change are increasing the pressures on ecosystems and ecological enhancement may provide a tool to provide suitable habitat for species through assisted colonisation (Hoegh-Guldbery et al., 2008).

It is important to carefully consider the rationale for ecological enhancement of artificial structures prior to creation and installation. For example, is the requirement as a compensation for habitat loss elsewhere in the region or are they primarily for an educational resource and local tourism? The interest shown by the general public at field events illustrates that these techniques can add value to these schemes by improving biodiversity and visitor engagement and awareness (Morris et al., 2016).

## 5 Conclusions

Increasing habitat heterogeneity on granite and limestone rock armour can promote and encourage biodiversity. The holes and grooves technique trialled here can be used at any stage of construction and are suitable for use in moderate and high wave energy environments where attached features such as tiles and artificial rockpools might not be suitable. In addition, the correct positioning of quarried boulders can also create habitats to maximise water-retaining features, for example where ‘blast lines’ or holes are already present. Future projects should upscale these smaller trials to large defence schemes, and aim to include a variety of sizes and depth of holes and grooves to further increase species richness and diversity of larger mobile species. Collaboration between ecologists and engineers is needed to develop multifunctional structures which can protect the land from coastal erosion and also create suitable habitat for marine organisms.

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