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Learning from nature to enhance Blue engineering of marine infrastructure

A.B. Bugnot*, M. Mayer-Pinto, E.L. Johnston, N. Schaefer, K.A. Dafforn

Evolution and Ecology Research Centre, School of Biological, Earth and Environmental Sciences, University of New South Wales, Kensington 2052, Australia

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ABSTRACT

The global sprawl of urban centres is replacing complex natural habitats with relatively flat and featureless infrastructure that supports low biodiversity. In a growing countermovement, artificial microhabitats are increasingly incorporated into designs for "Green" and "Blue" infrastructure. In order to maximise the ecological value of such interventions, we need to inform the designs with observations from natural systems and existing Green and Blue infrastructure. Here, we focussed on water retaining features mimicking intertidal rock pools, as this is a widely used intervention in coastal ecosystems. Using a meta-analysis and a qualitative literature review, we compiled information on diversity and function of rock pools on natural rocky shores and built structures to assess the potential ecological benefits of water retaining microhabitats and the design metrics of rock pools that affect diversity and function. Our meta-analysis showed higher species richness in rock pools compared to emergent surfaces on built structures, but this was variable among locations. The qualitative review revealed that rock pools on both natural and artificial shores generally hosted species that were not present on emergent rock and can also host non-indigenous species, suggesting that the addition of these features can sometimes have unwanted consequences and local ecological knowledge is essential to implement successful interventions. Relationships between species richness and design metrics, such as height on shore, volume, surface area and depth of pool were taxa-specific. For example, results from the meta-analysis suggest that building larger, deeper pools could increase diversity of fish, but not benthic organisms. Finally, this study highlights major gaps in our understanding of how the addition of rock pools and design metrics influence diversity and the variables affecting the ecological functioning of rock pools. Based on the knowledge gathered so far, recommendations for managers are made and the need for future studies to add knowledge to expand these recommendations is discussed.

1. Introduction

Natural habitats are shrinking and fragmenting due to the addition of built infrastructure, e.g. buildings and roads, seawalls and breakwaters, leading to a significant decline in biodiversity and ecosystem services in urbanised areas (Airoldi et al., 2008; Alberti and Marzluff, 2004; Grimm et al., 2008; Sala et al., 2000). Targeted modifications in the design of built environments are critical for species conservation and the recovery of lost diversity and function (Bergen et al., 2001; Dafforn et al., 2015b). A common form of modification has been the addition of microhabitats, such as roost-boxes on buildings for birds and bats (e.g. Brittingham and Williams, 2000; Goldingay, 2009) and water-retaining features on foreshore structures that aim to mimic natural rock pools (e.g. Chapman and Blockley, 2009; Firth et al., 2014a). Due to economic, logistical and engineering limitations, past designs of these features have tended to be simplistic and do not reflect the variability in size, shape and structural complexity of natural habitats (e.g. Browne and Chapman, 2014; Chapman and Blockley, 2009; Evans et al., 2016; Firth et al., 2016). With the development of new technologies such as 3-Dimensional printing, our capacity to design successful "Blue" and "Green" engineered structures is only limited by our understanding of ecological systems. In an effort to exploit available ecological knowledge to make cost-effective decisions for future interventions, we conducted a systematic review and meta-analyses and synthesised the current understandings of water retaining features as a common intervention in coastal ecosystems. On the basis of this study, we provide recommendations for new designs and/or modifications of urban marine infrastructure to increase biodiversity and ecological function and minimise ecological impacts.

Urbanised coastal areas have come to resemble "grey islands", with many natural habitats replaced and/or fragmented by infrastructure built for protection from land erosion and flooding (Bulleri and Chapman, 2010). These structures occasionally become fouled and attract fish life, and have been considered analogous to natural rocky

* Corresponding author.
E-mail addresses: a.bugnot@unsw.edu.au (A.B. Bugnot), m.mayerpinto@unsw.edu.au (M. Mayer-Pinto), e.johnston@unsw.edu.au (E.L. Johnston), n.schaefer@unsw.edu.au (N. Schaefer), k.dafforn@unsw.edu.au (K.A. Dafforn).

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shores (Thompson et al., 2002). However, assemblages on built structures differ in species identity and composition to those found on natural rocky shores, being less diverse (Chapman, 2003; Chapman and Bulleri, 2003) and supporting more non-indigenous species (Airoldi and Bulleri, 2011; Connell, 2001). One of the reasons for these differences is that built structures are typically subjected to greater disturbances and stressors than natural rocky shores (e.g. Airoldi et al., 2005; Airoldi and Bulleri, 2011). Furthermore, there are structural dissimilarities between rocky shores and built structures (Glasby, 1999). For example, breakwaters usually lack the variety of crevices, pits and rock pools found on natural rocky shores (Aguilera et al., 2014). These "irregularities" on natural rock surfaces provide a variety of microhabitats, which are used as refuge from predation and ameliorate the effects of disturbances and daily environmental fluctuations due to tidal cycles in the intertidal zone (Bertness et al., 1981; Fairweather, 1988; Garrity, 1984; Gray and Hodgson, 1998; Sebens, 1991). In addition, built structures are made of materials foreign to natural environments, such as concrete, plastic and treated wood, which have been shown to affect settlement of organisms (e.g. Anderson and Underwood, 1994; Glasby, 2000). As coastlines continue to be developed and require increasing infrastructure to protect valuable assets (Asif and Muneer, 2007; Thompson et al., 2002), mitigation strategies to manage and reduce associated impacts are essential.

Blue engineering designs incorporating ecological goals and principals (e.g. increase fish diversity) and informed by ecological knowledge have the potential to provide valuable habitat in highly modified environments (Bergen et al., 2001; Chapman and Underwood, 2011). To make built structures more suitable for the colonisation and establishment of native species and increase diversity, several design modifications and enhancements have been proposed, aiming at increasing structural complexity and, therefore, diversity of microhabitats. On land, for example, past studies have evaluated the characteristics of artificial roosts that maximise occupancy by targeted species (e.g. Mering and Chambers, 2014). Most interventions in marine environments, however, aimed to increase overall diversity and ecosystem function (Chapman, 2003), rather than to benefit particular species (but see Martins et al., 2010).

The marine intervention that has been most widely applied is the addition of water retaining features, which aims to mimic natural rock pools (Browne and Chapman, 2014; Chapman and Blockley, 2009; Evans et al., 2016; Firth et al., 2016; Firth et al., 2014a,b). The idea behind this approach is that a greater ecological diversity and/or species exclusively found in these features, in comparison to emergent rock, would translate into an increase in the overall diversity at the level of the structure or site. Hence, the success of previous interventions has been assessed by comparing biodiversity and species composition within and outside water retaining features, as opposed to the use of control sites (as discussed in Chapman et al., 2017). Using this approach, previous studies have reported an increase in biodiversity at the structure level after interventions (e.g. Browne and Chapman, 2014; Evans et al., 2016). However, such pattern is not universal and past surveys and experiments in natural rock pools and water retaining features have found conflicting results in terms of diversity and/or have not discussed the presence of unique species in rock pools (e.g. Araujo et al., 2006; Pinn et al., 2005; Segovia-Rivera and Valdivia, 2016). The future addition of water retaining features to artificial structures therefore requires further investigation. Consequently, the first aim of this study was to test whether rock pools host greater diversity than emergent rock and compiled information about the presence of unique species in rock pools using a qualitative literature review and meta-analysis.

Past interventions have been constrained by economical, logistical and/or engineering limitations, resulting in simple designs (e.g. a concrete flowerpot with smooth surfaces, Browne and Chapman, 2014; cylinder-shaped pools drilled in the rock, Evans et al., 2016), placed at constant tidal heights. To achieve maximum outcomes with cost-effective applications, designs should closely mimic the natural rock pools that enhance the desired variables. Progress in Green and Blue engineering therefore needs to be informed by observations of natural systems and learn from past attempts. The second aim of this study was to investigate the physical characteristics of rock pools that can be manipulated to maximise the diversity, processes and functions supported by these features using a qualitative review and, when enough data was available, meta-analyses. Results from this study are summarised in a decision tree to guide managers considering the addition of water retaining features to built infrastructure.

2. Methods

2.1. Systematic qualitative review

We did a literature review in the Web of Science™ on studies that examined ecological parameters (e.g. diversity, abundances, biomass, animal behaviour, processes and various ecosystem function variables) of rock pools on natural rocky shores and built structures. The search was done using the search terms "pool*" AND ("tide*" OR "tidal*" OR "rock*") for the period 01/01/1900 to 22/03/2017. After excluding results from unrelated research areas, we found 1,852 articles. These were further filtered by title and abstract, excluding articles that did not study intertidal rock pools (e.g. freshwater pools). We also searched the reference list of each selected study to capture studies that had not been included in the initial searches or that had been published in journals not indexed in the database we searched.

To evaluate the potential for water retaining features to be designed for ecological benefits, we selected studies that evaluated the diversity (number and identity of species), processes (e.g. grazing and predation) and functioning (e.g. primary and secondary productivity) in rock pools, including those that described the effect of design metrics (size, shape and position of pools) on ecological variables. These included manipulative and observational experiments, on both natural rocky shores and built structures. As a result, 156 studies that were included in our qualitative review.

2.2. Meta-analyses

Thirty-two papers identified in our systematic review (described above) were selected for the meta-analyses, based on the following criteria. Contrasts between studies require comparable methodologies and therefore studies were included in the meta-analyses only if they representatively sampled the local assemblages (i.e. by sampling all the organisms in a benthic quadrat or all fish collected by hand net). Studies that sampled a single or few taxa or species (e.g. Jorger et al., 2008; Schreider et al., 2003) were therefore excluded. Only studies that assessed 'established' or 'mature' assemblages were included in the meta-analyses. This was done to avoid potential confounding factors related to the stage of development of the assemblage. Therefore, for the purpose of this study, we defined "mature" assemblages as those older than 1 year. Studies where critical information (raw data, or mean, number of replicates and standard error or deviation) was missing were also excluded, due to statistical reasons. To standardise differences in sampling procedures (i.e. standardised sampling effort vs non-standardised), we estimated sampling effort as the volume of the rock pool sampled for fish and the area sampled for benthic assemblages. Thus, studies that did not contain sufficient information to calculate sampling effort were excluded from the meta-analysis.

Meta-analyses were done for a specific variable if there were more than 15 relevant data points from at least 6 studies (study identity could then be considered as a random factor – see data analysis for more details; Zuur et al., 2009). However, meta-analyses were also employed in cases where 2 or more studies reported raw data but had not tested the hypotheses addressed here. Limitations in such cases are, however, discussed. As a result, only studies reporting number of taxa for fish and benthic assemblages were included. Very few studies have measured ecological processes and function (e.g. productivity, grazing, predation, see results), therefore these variables were not investigated using meta-analyses. The comparison in number of taxa between rock pools and emergent rock and the relationships between design metrics tidal height, depth and size of rock pools with number of taxa were all analysed using meta-analyses. Other design metrics, including substrate complexity, incline and material, distance between pools and light incidence have been poorly defined and studied in natural rocky shores, and never evaluated from built infrastructure. They were therefore only assessed here using a qualitative review of the literature.

Data extraction from graphs was done using WebPlotDigitaliser (arohatgi.info/WebPlotDigitizer). We also extracted the location (name of the district/area) of each study, latitude, the type of structure (natural rocky shore vs built structure) and the total surface area (for benthic assemblages) or volume (for fish) of the sampling unit. Benthic assemblages and fish were analysed separately and sampling effort was included as a random variable in the models evaluating the effect of tidal height and depth. When evaluating the effect of area and volume for benthic assemblages, standardised and not-standardised studies were evaluated separately, resulting in two analyses per design metric.

When comparing number of taxa between rock pools and emergent rock, some studies reported mean and errors per location sampled, while others reported those for different sites within a location (e.g. several groynes in the same location used by Pinn et al., 2005). Each site or location was considered a replicate, hence several data points per study were included in the meta-analyses. A multi-level model with "number of taxa" as response variable and "habitat" (artificial vs natural) as explanatory fixed factor was done using rma.mv (*metaphor* package in R 3.3.3). To account for non-independence in the data, "study identity" and "location" (nested in study identity) were included as random factors (Noble et al., 2017). The effect size of number of taxa between rock pools and emergent rock was calculated using the log transformed ratio of means (using "ROM" in *metaphor* package in R). Sensitivity analyses were performed using the leave-one-out method, by removing one study or one location at the time to detect data points with strong effects in the results (Borenstein et al., 2009).

Studies reporting the effects of design metrics on number of taxa used a mix of raw data and means and errors per site. Hence, a logarithmic transformation was used to control for the differences in the distribution of variances. To assess the relationships between design metrics and number of taxa, linear models were used (*nlme* package in R) with "number of taxa" as response variable, the design metrics (tidal height, depth, volume or area) as explanatory variable and including "study identity" and "location" as random predictors to account for non-independence (Nakagawa and Santos, 2012; Noble et al., 2017). For analyses where the level of the factor "study identity" was lower than 6, this was considered a fixed factor (Zuur et al., 2009) and the analyses were done using a linear model ('lm' function in R). The number of replicates per data point (1 for raw data points, > 1 for means) was included as *weights* in all models. For benthic assemblages, tidal height was reported as a categorical and/or continuous variable, so two separate analyses were performed to test for consistency in the effects of tidal height. When evaluating the relationships between number of taxa and size of rock pools, area and volume were log transformed, as the species-area relationship predicts a log-log relationship (Connor and McCoy, 1979).

3. Results

Form the 156 studies included in our qualitative review, 32 of them directly compared ecological parameters between rock pools and emergent rock, while of those, 12 assessed the diversity and composition of assemblages. Twenty-five out of 32 articles focussed on natural rocky shores, while 7 focussed on built infrastructure. Design metrics were evaluated in 133 studies, with tidal height being the variable most

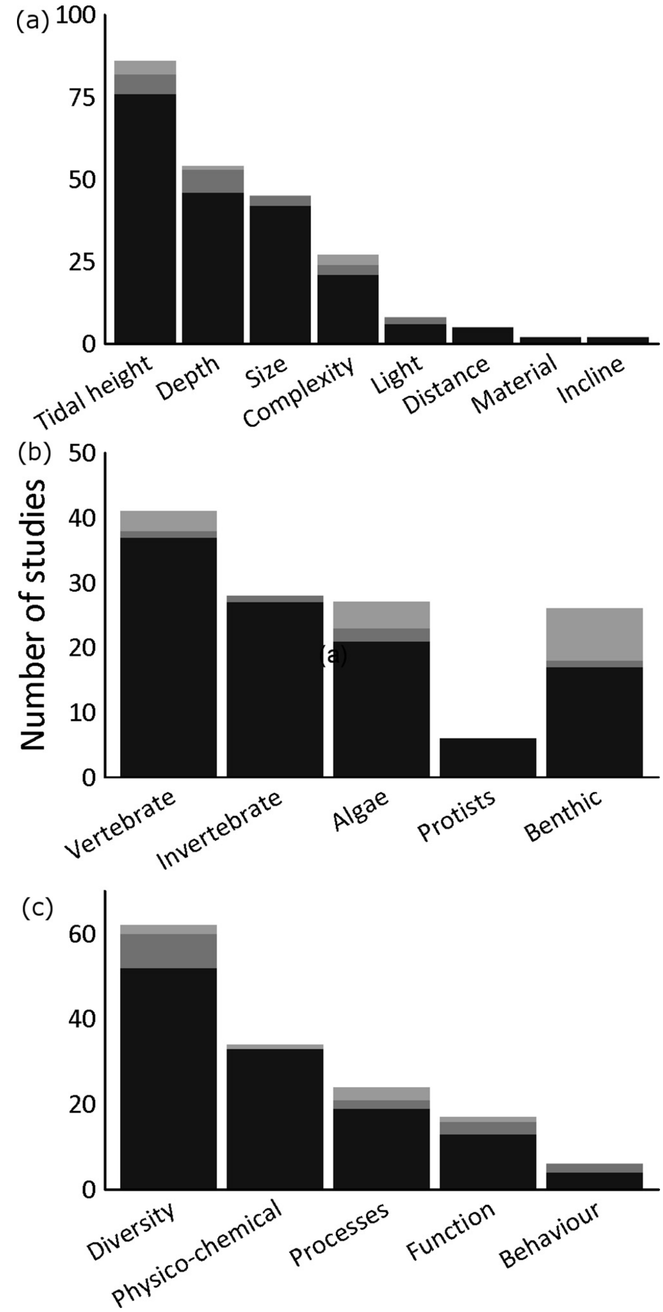

Fig. 1. Number of studies that assessed physical and ecological parameters in intertidal rock pools on natural rocky shores (black), built infrastructure (dark grey) or both (light grey). (a) Studies evaluating design metrics (tidal height, depth, size, structural complexity (complexity), light incidence (light), distance away from other pools (distance), substrate incline and construction material (material)). (b) Studies focused on vertebrates, invertebrates, algae, protists and all benthic organisms. (c) Studies assessing community diversity (including diversity parameters such as Shannon Index and total number of taxa, total abundances, biomass and multivariate analyses), physico-chemical characteristics of the pools (temperature, pH, salinity, oxygen, etc.), ecological processes (reproduction, grazing, predation, connectivity, dispersal), ecological functioning (production, respiration, nutrients cycling) and behaviour.

studied, followed by depth, size (reported as volume and/or area), structural complexity, light incidence, distance between pools, material and incline (Fig. 1a). Vertebrates (mainly fish) were the taxa most studied, followed by invertebrates and algae, while a great number of studies analysed benthic assemblages, which included invertebrates and algae (Fig. 1b). The ecological variables most frequently reported were biodiversity parameters, such as number of taxa and abundance/cover of organisms (Fig. 1c). Other variables reported included physical variables such as temperature, salinity and pH and ecological processes such as recruitment, grazing and predation. Functioning and behaviour were poorly studied, with only 17 and 6 studies respectively (Fig. 1). The presence and/or distribution of non-indigenous species in rock pools was only evaluated in 7 studies.

Twenty-two studies from the 156 identified in our systematic review were selected for the meta-analyses (Supplementary Material 1). These included 16 studies on natural rock shores, 5 on built structures and 1 in both natural and built habitats. All studies comparing the number of taxa between rock pools and emergent rock focussed on benthic assemblages, whereas studies that evaluated the relationships between design metrics and number of taxa focussed on benthic and fish assemblages.

3.1. Diversity, processes and function in rock pools vs emergent rock

Using a meta-analysis, we compared the number of taxa between rock pools and emergent rock at 38 sites (each a replicate) included in 21 locations extending over 20 degrees in latitude in the European Atlantic Ocean, and gathered from 6 studies (Fig. 2, Supplementary Material 1). These studies were done on natural rocky shores (19 sites), breakwaters (18 sites) and one site on a Bioblock®. The great majority were observational, with only one manipulative experiment.

Although most individual studies comparing diversity in pools with emergent rock found greater number of species in rock pools (but see Pinn et al., 2005), our meta-analysis revealed that this pattern is not consistent in space, as we found great variability in the effect sizes between sites (Fig. 2; Test for Heterogeneity Qe = 138.55, df = 36, $p < 0.0001$). In addition, results from the meta-analyses showed that number of benthic taxa was more abundant in water-retaining features than on the emergent surface only on built structures ($z = 2.76$, $n = 19$, $p = 0.0058$, Fig. 2) and no pattern was observed in natural rocky shores ($z = 1.38$, $n = 19$, $p = 0.17$, Fig. 2). Additionally, our analyses showed that a single location from a single study (Firth et al., 2013, which includes the location Elmer, among others) was the main driver for the differences found between pools and emergent substrata on artificial habitats, as the exclusion of this location or study significantly changed the interpretation of results (i.e. no differences in diversity between pools and emergent substrata; Supplementary Material 2).

Our qualitative literature review, on the other hand, showed that regardless of the total number of taxa inside vs outside the pools, rock pools can provide habitat for a variety of species not found on emergent rock. For example, a rocky shore study in Portugal observed two algal species that were unique to rock pools, but overall a greater number of taxa were found on emergent rocks than in rock pools (Araujo et al., 2006). Lintas and Seed (1994) also found a greater number of taxa in mussel beds on emergent rock compared to rock pools on natural shores, but six species were exclusively found in rock pools (an anemone, a polychaete worm, a gastropod, a bryozoan and two species of copepods). Similar patterns were found for built structures, where water retaining features were colonised by species not found on the emergent surface. For example, on the high shore of seawalls in Sydney Harbour, ascidians and sponges were only found in constructed rock pools (Chapman and Blockley, 2009). Evans et al. (2016) found that

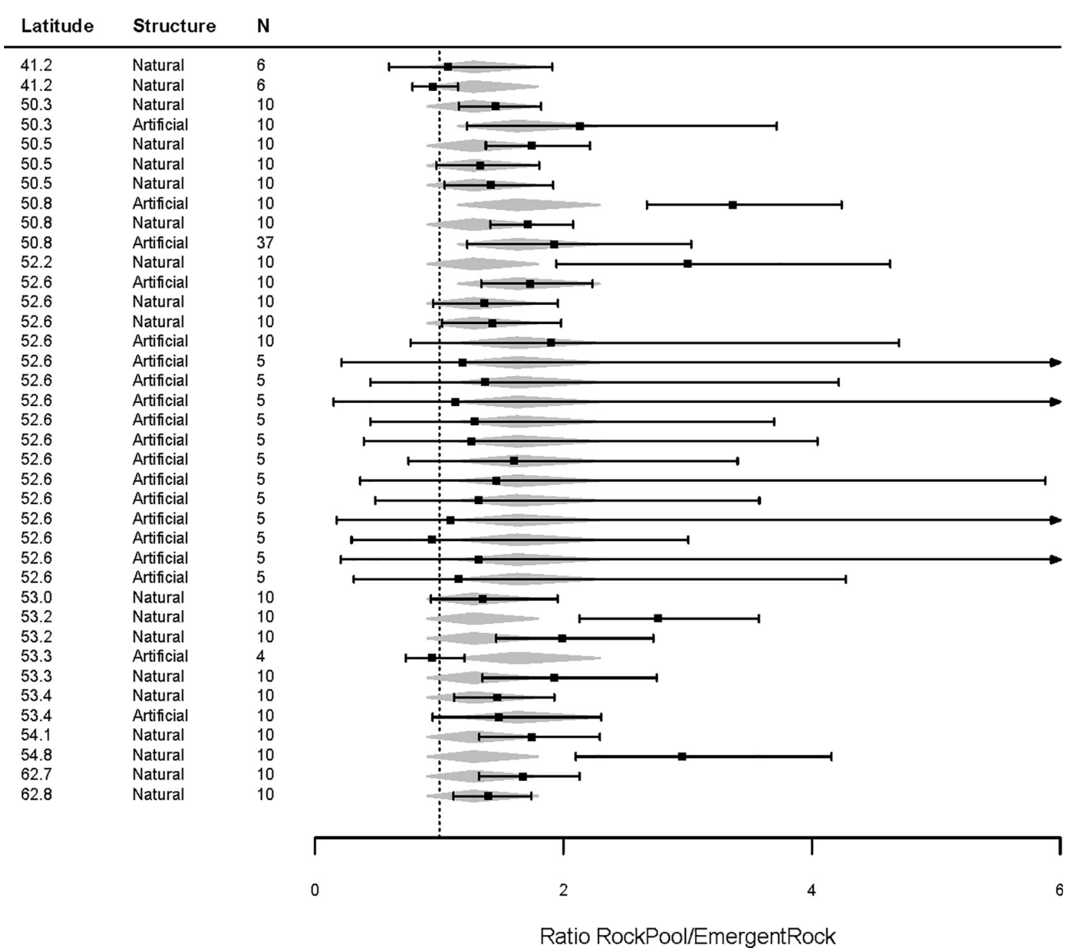

Fig. 2. Ratio between number of taxa in rock pools vs emergent rock for each data point. Grey diamonds represent the estimated value and 95% CI for each group (built structures and natural rocky shore). Data points are ordered by latitudinal gradient. N = number of replicates.

adding rock pools to a breakwater supported groups completely absent from emergent surfaces, such as fish, ascidians, bryozoans, hydroids and sponges. The generality of this pattern is, however, still uncertain as some studies have not reported species uniquely found on rock pools (e.g. Pinn et al., 2005; Segovia-Rivera and Valdivia, 2016). In addition, rock pools were observed to benefit non-indigenous species. Andrew and Viejo (1998) found a greater prevalence of the non-indigenous algae *Sargassum muticum* in rock pools than on emergent rock in Spain.

Our qualitative review also showed that rock pools have different functional characteristics and processes than emergent rock. Grazing intensity was two times greater in rock pools than on emergent rock in natural rocky shores of England, due to a species of limpet that often migrated to rock pools for feeding purposes (Noel et al., 2009). Therefore, organisms not usually reported in the pools might be directly or indirectly benefiting from the presence of these habitats.

3.2. Tidal height

A total of 26 and 52 data points from 3 and 4 studies were included in the meta-analyses to assess the relationships between tidal height and fish and benthic diversity, respectively, using tidal height as a continuous variable. Additional 49 data points from 8 studies were used to test this relationship for benthic assemblages using tidal height as a categorical variable (Supplementary Material 1).

The meta-analyses showed that number of taxa for fish and benthic assemblages were negatively related to tidal height (Table 1, Fig. 3). Our qualitative review showed that species composition also changed with tidal height. For example, some fish species occurred in rock pools at all tidal heights (e.g. *Clinocottus analis* in California and *Blennius pholis* in the North coast of France), while others were present only at some heights (e.g. *Girella nigncans* was absent from lower pools in California, while *Coryphoblennius galerita* and *Gobius cobitis* were limited to higher pools in the North coast of France; Davis, 2000; Gibson, 1972). Dominance of algal groups in rock pools was also related to tidal height (Araujo et al., 2006; Green, 1971). Huggett and Griffiths (1986) found that rock pools located at the low shore were dominated by bivalves and sponges, whereas pools at the high shore were dominated by algae and grazers.

Functioning of rock pools was also reported to vary with tidal height, but the direction of these relationships varied depending on the species. Primary productivity was found to be greatest in pools at mid-heights in Portugal (Alvera-Azcarate et al., 2003) and at the lowest pools in the United Kingdom (Martins et al., 2007). Annual production of fish was also observed to vary with tidal height, but the direction of the relationship varied for different species (Mgaya, 1992). Growth of an invasive algae *Codium fragile* was greater in lower pools in Canada (Scheibling and Melady, 2008).

3.3. Depth

The meta-analyses testing the relationships between depth of pools and number of species of fish included 69 data points from a total of 7 studies. For benthic assemblages, we analysed a total of 27 data points from 3 studies (Supplementary Material 1). The analysed depths of rock pools ranged from less than 1 cm to 2 m. We found a significant positive relationship between depth of pool and fish diversity ($Chi^2 = 15.06$,

Table 1
Results from the linear models assessing the effect of tidal height on number of taxa, when using tidal height as a continuous and categorical variable.

	Continuous			Categorical	
	Estimate	Chi^2	p	Chi^2	p
Fish	−0.50 ± 0.17	15.04	0.001		
Benthic	−0.65 ± 0.28	43.49	0.0000	26.54	0.0000

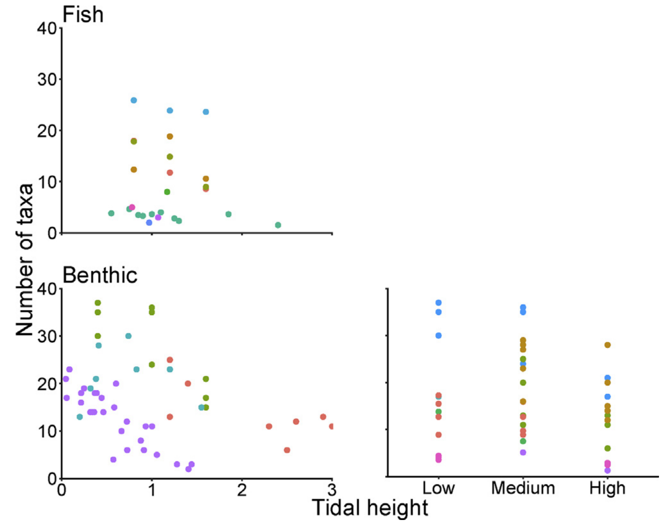

Fig. 3. Number of taxa vs tidal height of rock pools for fish (top) and benthic (bottom) assemblages. For benthic assemblages, studies using tidal height as a continuous variable (meters) are on the left, and using a categorical variable are on the right. Colours represent different sites.

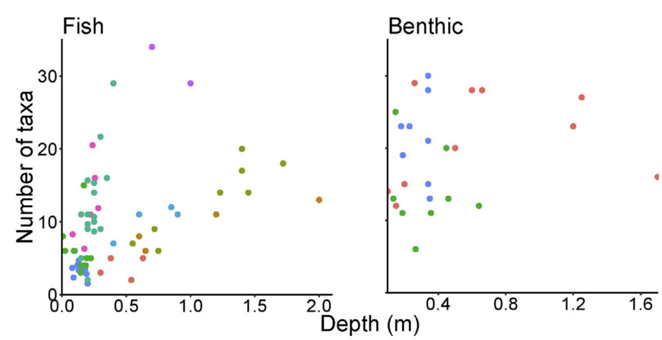

Fig. 4. Number of taxa vs depth of pool (meters) for fish and benthic assemblages. Colours represent different sites.

$p = 0.002$, Fig. 4). In contrast, the number of taxa of benthic assemblages was not related to depth ($F = 0.42$, $p = 0.52$, Fig. 4).

These results may be explained by our qualitative review that showed that such relationships are species-dependent. For instance, a study done in Australia, with rock pools ranging from 5 to 40-cm deep, found that the grazer snail *Cellana tramoserica* was more abundant in deeper pools, whereas the algae *Hilderbrandia prototypus* was more abundant in shallower pools (Astles, 1993). Tubeworms, molluscs and crustaceans were observed to colonise shallow (22 cm) water retaining features added to seawalls in Sydney, while opportunistic algae were dominant in deeper (38 cm) pools (Browne and Chapman, 2014). In addition, the distribution and abundances of the non-indigenous gastropod *Zeacumantus subcarinatus* were also affected by depth, but the patterns observed were complex. Densities of *Z. subcarinatus* were negatively correlated with depth only when considering pools where it was present (Hendrickx et al., 2015). In contrast, a manipulative experiment where built rock pools were added to a breakwater found no effect of depth on the composition of assemblages (Evans et al., 2016).

Our qualitative review also highlighted possible relationships between functioning and depth of pools. Primary productivity varied with depth in pools from 2 to 24 cm deep, with deeper pools having greater productivity when located on the upper shore (Martins et al., 2007). The relationship between depth (range of depth studied not reported) and growth of two fish species varied depending on the species, with one species, *Clinocottus globiceps*, showing no significant trends and another, *Oligocottus maculosus*, showing a negative relationship (Mgaya,

Table 2
Results from the linear models assessing the effect of size (measured as volume and area) on number of taxa for fish and benthic assemblages.

	Volume			Area		
Not standardised	Estimate	Estimate	p	Estimate	Estimate	p
Fish	1.21 ± 1.05	$Chi^2 = 46.96$	0.0000	1.31 ± 1.04	$Chi^2 = 51.00$	0.000
Benthic		$F = 2.08$	0.17		$F = 2.13$	0.17
Standardised		t value	p		t value	p
Benthic		$F = 1.75$	0.21		$F = 1.45$	0.25

1992).

3.4. Size (volume and area)

The relationship between number of fish species and volume was assessed using a meta-analysis with a total of 73 data points from 8 studies, whereas analyses testing fish diversity and area had a total of 69 data points from 7 studies (Supplementary Material 1). The meta-analysis for benthic taxa and size (both volume and area) of rock pools was done with a total of 18 data points from 2 studies, all with standardised sampling (Goss-Custard et al., 1979; Metaxas et al., 1994). In addition, we analysed a further 16 data points from 2 studies that used non-standardised sampling (Underwood and Skilleter, 1996; Wolfe and Harlin, 1988). Area of pools analysed ranged from 7.1 to 876 m^2 and volume from 0.0003 to 14300 m^3.

The meta-analysis found that number of fish species had a significant, positive log-log relationship with both volume and area (Table 2, Fig. 5), as expected based on the richness-area relationship (Connor and McCoy, 1979). However, the number of taxa of benthic organisms did not show a significant relationship with volume or area for standardised and non-standardised data (Table 2, Fig. 5). Similar to our results on depth, this discrepancy between fish and benthic assemblages are likely due to a greater variation in the responses of benthic species to pool size as highlighted by our qualitative review. The review showed that the effects of rock pool size on assemblage composition of benthic organisms varied in space and time (Bussell et al., 2007; Jordaan et al., 2011; Martins et al., 2007), as well as with species identity (Zhuang, 2006).

The relationship between productivity and size was evaluated in two studies. Primary productivity was found to be positively correlated with pool volume, but not with area (for pools ranging from 0.03 to 1.9 m^2, Martins et al., 2007), while annual production of fish was not related to area or perimeter of rock pools (sizes of pools samples not reported, Mgaya, 1992).

3.5. Other variables

Our qualitative review found 27 studies that evaluated complexity of the substratum and implemented a range of definitions for complexity, from surface rugosity (e.g. Daryanavard et al., 2015; Macieira and Joyeux, 2011), presence of rock, rubble and sand (e.g. Griffiths et al., 2006; Mahon and Mahon, 1994), to presence of pits and crevices (e.g. Cunha et al., 2008; Dumas and Witman, 1993). Studies evaluating the effects of surface complexity (e.g. rugosity, small pits) on a series of ecological variables, such as diversity (e.g. Davidson and Grupe, 2015; Griffiths et al., 2006), species composition (e.g. Daryanavard et al., 2015; Matias, 2013), behaviour (Mayr and Berger, 1992; Nakamura, 1976) and primary production (Matias, 2013), among others, found varying results. For example, the relationship between densities of fish and rugosity differed between species in a study in California (Davis, 2000). Griffiths et al. (2006) found that the addition of rocks (sizes not reported) to natural pools resulted in an increase in number of species and abundances of fish, but no effect was observed on size distribution of fish among pools. In contrast, abundances of fish were only weakly related to the percentage cover of rocks and no relationship was observed with number of species and biomass in South Africa (Marsh et al., 1978). Algal assemblages, however, were negatively affected by the presence of rocks (5–8 cm in diameter) due to a scouring effect (VanTamlen, 1996).

The metrics light incidence, distance between pools, material and incline of the substratum inside pools have been poorly assessed before (8, 5, 2 and 2 studies, respectively), and studies have evaluated a range of response variables. The effect of light incidence on ecological communities in natural rock pools has been poorly studied and was never evaluated on water retaining features on built structures. Previous studies have shown that shaded rock pools have different algal assemblages than those that are illuminated the whole day, with light favouring Cyanophytae and Chlorophyceae at the expense of Rhodophyceae and Phaeophyceae (Gustavss, 1972). However, UV exposure can also have negative effects on algae and fish. Coralline algal species showed photo-inhibition related to irradiance fluctuations in rock pools (Williamson et al., 2014). The fish *Girella laevifrons* also showed signs of stress when exposed to UV radiation, by increasing oxygen production, decreasing body weight and actively searching for refuge (Pulgar et al., 2017).

Distances between rock pools showed variable effects on population dynamics, however no studies evaluated their effect on assemblage composition. When evaluating temporal changes in population dynamics of a copepod, pools between 0 and 40 m apart showed similar patterns of temporal variation, but pools more than 40 m away showed different patterns of temporal variation (Johnson, 2001). Population synchrony of an invertebrate species was also negatively related to distance among pools (Pandit et al., 2016). In contrast, Engel et al. (2004) found that genetic differences of the red seaweed *Gracilaria gracilis* between pools were not related to distance between pools. Only one study assessed the effects of material of rock pools on community structure. Cox et al. (2011) sampled rock pools on basalt and limestone natural rocky shores and found different fish assemblages. No studies have, however, evaluated the effect of material of construction when designing water retaining features for foreshore infrastructure. Incline of the pool can influence diversity and functioning as it affects shading and depth, but only one study evaluated this factor. Firth et al. (2014a) reported that the relationship between incline and richness varied with functional group, with canopy algae having a positive relationship with slope, while encrusting algae and faunal groups had no clear trends.

4. Discussion and recommendations

Green and Blue engineering strategies are increasingly being recognised as valuable tools to enhance biodiversity and function in highly modified environments (Dafforn et al., 2015b; Schifman et al. (2017); Threlfall et al., 2017). Their success and applicability are, however, limited by our understanding of the particular factors driving biodiversity and function. For example, if the objective of an intervention is to maximise ecological biodiversity on a particular structure through the addition of microhabitats, what are the specific

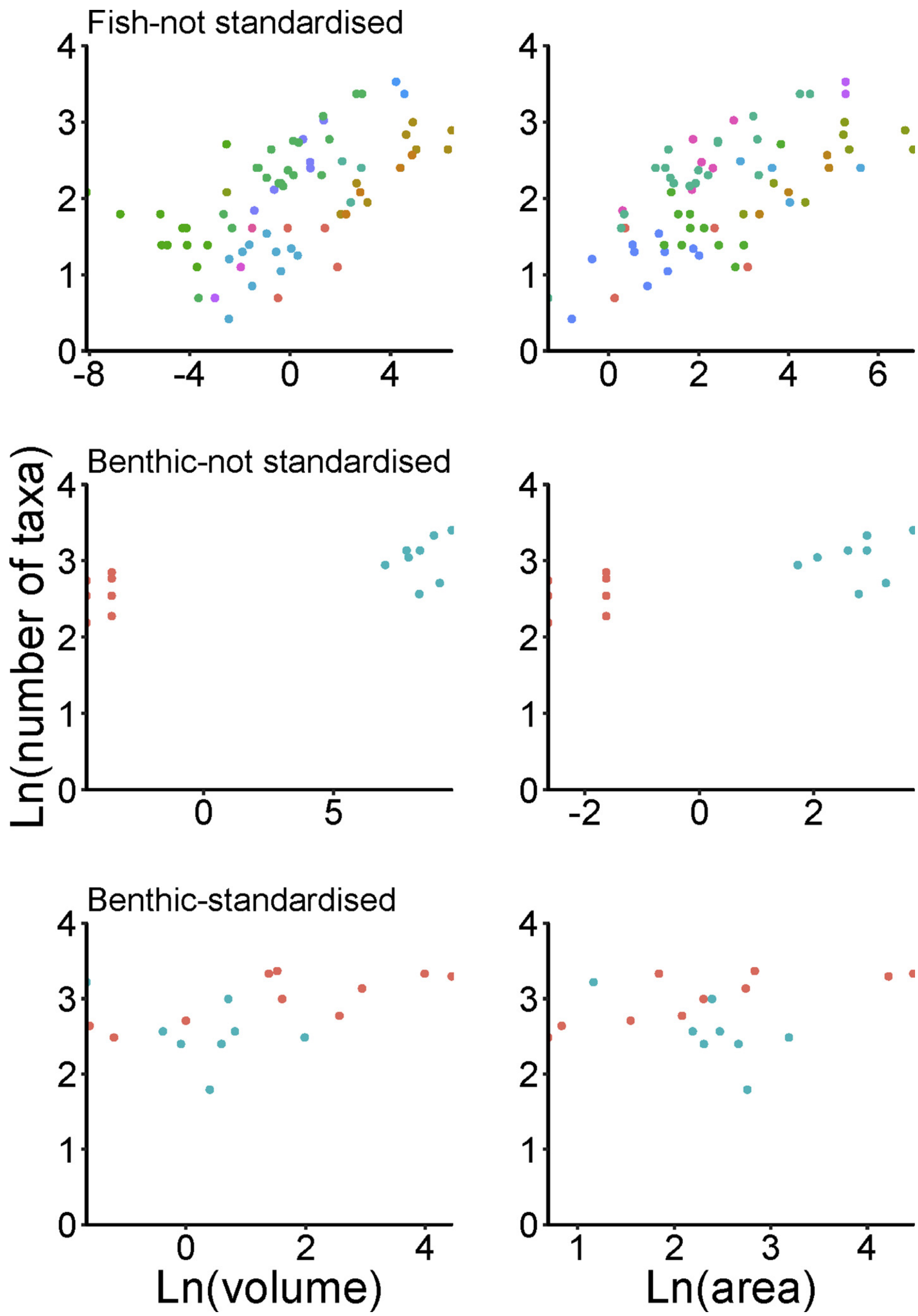

Fig. 5. Number of taxa vs size (volume and area, log-log relationship) for fish and benthic assemblages. Studies using standardised (only for benthic) and not-standardised sample size. Colours represent different sites. (For interpretation of the references to colour in this figure caption, the reader is referred to the web version of this article.)

characteristics of these microhabitats that maximise biodiversity? This study takes the first steps into answering these questions by gathering the existing knowledge on the effects of the presence and characteristics of rock pools on biodiversity and function. Using a combination of qualitative literature review and meta-analyses, we propose the first guidelines for the design of water retaining features on built infrastructure (Fig. 6), which should be updated and expanded as new data becomes available.

As evidenced by our meta-analyses and qualitative review, Blue engineering strategies have the potential to add a greater range of species to built infrastructure, but should be informed by knowledge of the local environment, flora, and fauna rather than following the general assumption that adding habitat complexity to artificial structures will increase native species diversity (Fig. 6). Our meta-analysis revealed, that contrary to the general perception that rock pools contain greater diversity than emergent rock, these patterns are variable in space. Nevertheless, our in-depth qualitative review showed that water retaining features can host a set of species absent on the surrounding emergent rock, although the generality of this idea is still uncertain, as some studies have not reported species uniquely found in rock pools. Whether this was a case of under-reporting or simply represents a lack of unique species in rock pools is unclear. In addition, care should be taken not to facilitate the establishment and/or spread of noxious, invasive species (Fig. 6, Dafforn, 2017).

Our analyses testing the relationships between tidal height and diversity of rock pools showed that tidal height (in both built structures and natural habitats) is negatively related to biodiversity of fish and benthic assemblages. In addition, the qualitative review also showed

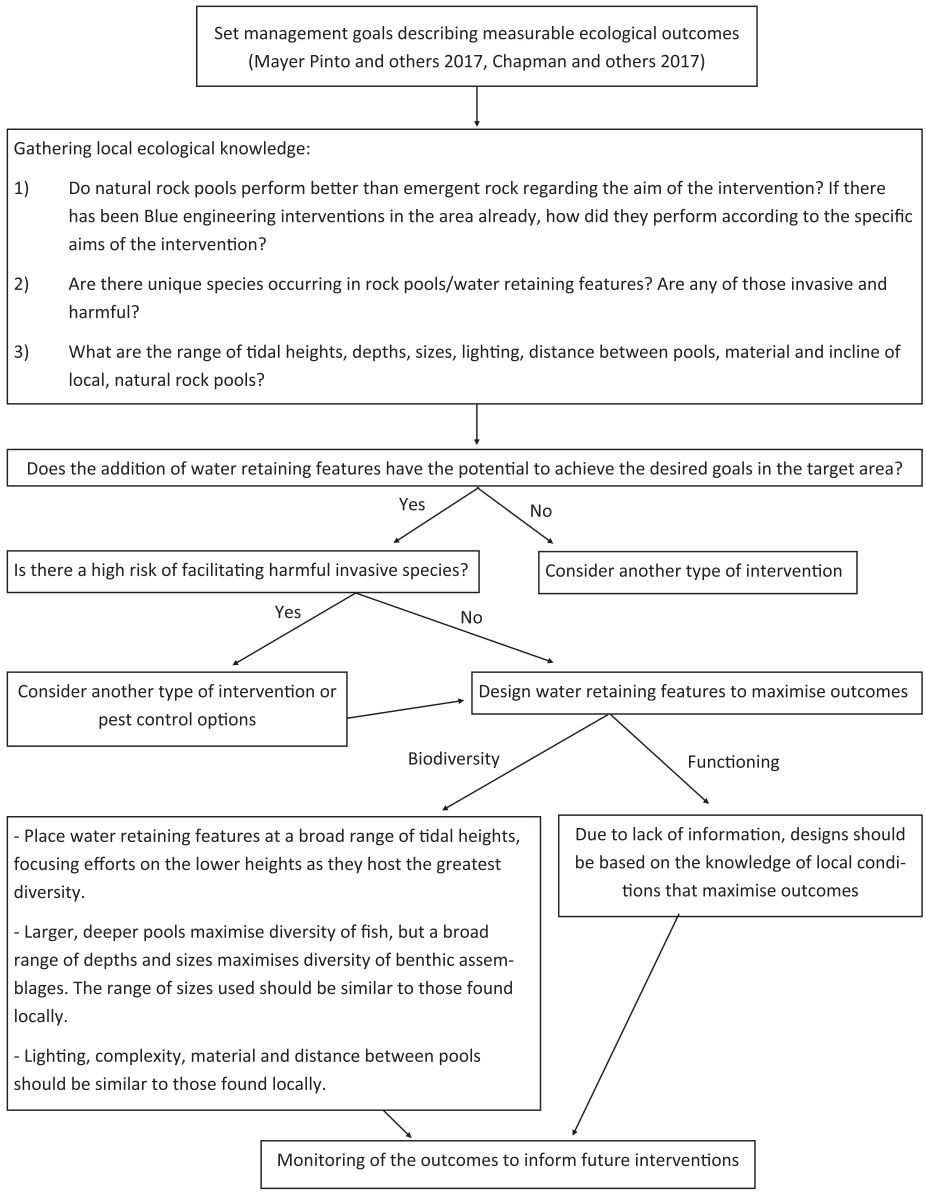

Fig. 6. Proposed decision tree based on the results of the meta-analyses and qualitative review for evaluating the appropriateness of the addition of water retaining features and decide on its design metrics.

evidence of changes in function with tidal height. For example, even though greater biodiversity was always observed in pools in the lower intertidal compared to pools in the higher intertidal, primary productivity peaked at a range of tidal elevations, suggesting context dependant relationships between these two variables. When designing Blue infrastructure that incorporates water retaining features, tidal elevation is an important parameter to consider and should be selected based on the desired outcomes of the intervention (e.g. to optimise diversity or function or both). The results of this study, therefore, suggest that in cases where the main objective is to increase diversity at the site level, water retaining features should be placed at a broad range of tidal heights to provide suitable habitat to a wide variety of species (Fig. 6). If resources are constrained, the majority of water retaining features might be placed in the low intertidal zone, to maximise habitat availability in this high-diversity zone. If, on the other hand, the main objective is to increase a particular function of the system, e.g. productivity, the tidal height of these features will be site dependent and developers might choose the most suitable elevation range for the dominant local keystone species (Fig. 6).

Our meta-analyses revealed that deeper, larger pools increased the diversity of fish, however, these results were not reflected in the diversity of the benthic assemblages. These patterns can be explained by the requirements of highly mobile fish (e.g. Davis (2001); Jordaan et al., 2011) and behavioural choices (e.g. Davis, 2000; Richkus, 1981). The lack of trends observed for benthic assemblages might be due to such relationships being species-dependent, as per suggested in our qualitative analyses. Alternatively, this could be due to the low number of locations and studies included in this analysis, as most studies have not reported data to standardise for sampling effort (see methods). Therefore, based on the information available to date, we recommend that, if the main objective is to increase fish diversity, efforts should be concentrated on building larger, deeper pools (Fig. 6). If, however, the main goal of a modification is to increase overall diversity, water retaining features with a broad range of depths, sizes and structural complexity should be added to the habitat to increase niche variability at the site (Fig. 6). Finally, the relationships between depth and size of pools and functioning were observed to vary greatly between and within studies, so any designs aiming to maximise particular functional attributes should be based on local ecological knowledge (Fig. 6).

In polluted environments in particular, care should be taken to avoid the creation of contaminant traps by varying the pool height to width ratio (e.g. a 3:1 height to volume ratio is an efficient particle trap and may result in contaminated water features, Dafforn et al., 2013). Also, shape and size of pools can affect physico-chemical characteristics such as temperature (Kita et al., 1985; Martins et al., 2007) and desiccation (Altermatt et al., 2012), which can, in turn, affect colonisation and establishment of communities. Therefore, designs should aim to minimise these effects by using a range of depth and sizes similar to those of local natural rock pools (Fig. 6). Additionally, maximum outcomes can only be obtained if Blue engineering strategies are implemented in conjunction with measures to control disturbances on these structures, including the implementation of maintenance practices that minimise physical damage to ecological assemblages and discourage harvesting of native organisms (Airoldi et al., 2005; Airoldi and Bulleri 2011).

Even though the number of studies is relatively low for some of the analyses done here, our meta-analyses have revealed different patterns than individual studies have previously shown. By combining data from a variety of studies done at different times and places, we were able to, not only reveal more general patterns (or the lack of), but also ask different questions than those from the original studies. For instance, some of the studies that provided raw data to our analyses described temporal or spatial patterns (Goss-Custard et al., 1979; Metaxas et al., 1994), or discussed changes in community composition (Daryanavard et al., 2015; Wolfe and Harlin, 1988), but did not formally tested the relationships described here. In addition, our meta-analyses included sampling effort in the models to standardise the data, which had not been done by previous studies (e.g. Firth et al., 2013; Macieira and Joyeux, 2011; Wolfe and Harlin, 1988). This is imperative to obtain an unconfounded relationship between diversity and the parameters tested here.

Our qualitative review revealed that certain design metrics remain almost unexplored. Marine infrastructure is greatly affected by anthropogenic sources of light, such as artificial lighting at night and shading caused by infrastructure during the day. On pilings and pontoons, artificial light affects fish abundance and predation on benthic assemblages (Bolton et al., 2017), while daylight intensity and surface inclination influence the presence and competitive dominance of non-indigenous species (Dafforn et al., 2012). The effect of light on ecological communities in natural rock pools has been largely overlooked and has yet to be evaluated on water retaining features on built structures. Inclination of rock pools and its effect on associated assemblages represents another big gap in our understanding, particularly given the demonstrated importance of surface orientation for benthic assemblages (Blockley and Chapman, 2006; Dafforn et al., 2012; Glasby, 2000). Finally, construction material is understood to affect benthic assemblages (e.g. Anderson and Underwood, 1994; Brown, 2005; Burt et al., 2009; Hawkins et al., 2010), but only one study has examined its effect on rock pool communities (Cox et al., 2011). We strongly recommend the measurement of such parameters in future studies to inform the design of water retaining features, as well as that of marine infrastructure more generally. In the absence of such information for pools, designs of these features should minimise artificial light, maximise natural light, and as far as possible mimic the surface characteristics of the local coastal geology (Fig. 6).

Even though there is a broad number of studies focussing on the ecology of rock pools, the community-level effects of the presence of rock pools and their design metrics remain poorly studied. In addition, previous interventions are limited to small scales, whereas the effects of interventions at the scale of the site remain unexplored (Chapman et al., 2017). Furthermore, as in studies on human impacts on ecological communities (Johnston et al., 2015), rock pool studies tend to be focussed on community structure and little is known about impacts on the ecosystem functions that underpin critical services. Our review uncovered case studies showing that rock pools on natural rocky shores can affect the functioning of communities and can have bottom-up effects that escalate, at least, to the whole habitat. One example from a natural rocky shore at the coast of the United Kingdom, described how limpets migrated from neighbouring emergent rock to graze inside rock pools (Noel et al., 2009). As Blue engineering interventions can serve multiple biodiversity and functional objectives (Dafforn et al., 2015a) and indirect assessment of functioning using diversity measures is not accurate (Johnston et al., 2015), we strongly recommend including direct measurements of functional variables in future studies. In the face of scarce information, designs aiming to increase a functional aspect of the habitat should be informed by the drivers of the functional parameter to maximise (Fig. 6). To build on the available knowledge, fill knowledge gaps and improve the recommendations proposed here, future interventions should design appropriate monitoring strategies (as discussed by Chapman et al., 2017) and make the data publicly available.

Even when achieving well tested and thorough recommendations, the results of these interventions cannot be guaranteed, as the local pool of species can greatly influence the outcomes of future interventions. For example, this review has found that interventions on foreshore infrastructure may also facilitate the establishment of non-indigenous species. This has also been observed in terrestrial environments where artificial roots have been colonised by non-indigenous birds, insects and mammals (e.g. Le Roux et al., 2016; Savard and Falls, 1981). With the objective of pre-empting space and avoiding the settlement of undesired species, a recent study added artificial turf to water retaining features (Morris et al., 2017). Results showed no differences in the percentage on non-indigenous species between pools with and without turf. Hence, local knowledge is imperative to forecast issues and prepare appropriate management strategies to maximise results (Fig. 6).

5. Conclusions

The fast advance of design and construction technology provides new opportunities to improve and refine ecological interventions based on in-depth observations of natural systems and lessons learnt from existing Green or Blue infrastructure. Clear ecological objectives of the intervention, proposed *a priori*, are crucial to inform the most appropriate designs for infrastructure, a practice often neglected (Chapman et al., 2017; Mayer-Pinto et al., 2017). Notwithstanding the aforementioned knowledge gaps, this study produced recommendations that, in combination with general ecological understandings of intertidal ecosystems, can guide the design of foreshore constructions. The results of this review support the idea that the addition of microhabitats to built infrastructure has the potential to increase diversity and functioning, but local knowledge is crucial to design targeted interventions. Some key principles include: (a) managers should evaluate if the addition of water retaining features has the potential to achieve the desired outcomes based on knowledge of the local ecological communities, (b) to achieve benthic biodiversity goals, niche diversity should be maximised by adding water retaining features of varying tidal height, depth, pool volume and area; whereas to increase fish biodiversity, deeper bigger pools should be prioritised, (c) aim for natural dimensions, materials, complexity, light regimes and distance between pools similar to those found locally. As we continue to construct the coast, we emphasize the need for small and large-scale studies that address critical knowledge gaps by doing rigorous monitoring, assessment and reporting of ecological outcomes (including open access for raw data) to build our knowledge base and continually update and expand the recommendations proposed here.

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Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at http://dx.doi.org/10.1016/j.ecoleng.2018.03.012.

References

Aguilera, M.A., Broitman, B.R., Thiel, M., 2014. Spatial variability in community composition on a granite breakwater versus natural rocky shores: lack of microhabitats suppresses intertidal biodiversity. Mar. Pollut. Bull. 87, 257–268.

Airoldi, L., Bacchiocchi, F., Cagliola, C., Bulleri, F., Abbiati, M., 2005. Impact of recreational harvesting on assemblages in artificial rocky habitats. Mar. Ecol. Prog. Ser. 299, 55–66.

Airoldi, L., Balata, D., Beck, M.W., 2008. The gray zone: relationships between habitat loss and marine diversity and their applications in conservation. J. Exp. Mar. Biol. Ecol. 366, 8–15.

Airoldi, L., Bulleri, F., 2011. Anthropogenic disturbance can determine the magnitude of opportunistic species responses on marine urban infrastructures. PLoS One 6.

Alberti, M., Marzluff, J.M., 2004. Ecological resilience in urban ecosystems: linking urban patterns to human and ecological functions. Urban Ecosyst. 7, 241–265.

Altermatt, F., Bieger, A., Morgan, S.G., 2012. Habitat characteristics and metapopulation dynamics of the copepod Tigriopus californicus. Mar. Ecol. Prog. Ser. 468, 85–93.

Alvera-Azcarate, A., Ferreira, J.G., Nunes, J.P., 2003. Modelling eutrophication in mesotidal and macrotidal estuaries. The role of intertidal seaweeds. Estuarine Coast. Shelf Sci. 57, 715–724.

Anderson, M.J., Underwood, A.J., 1994. Effects of substratum on the recruitment and development of an intertidal estuarine fouling assemblage. J. Exp. Mar. Biol. Ecol. 184, 217–236.

Andrew, N.L., Viejo, R.M., 1998. Ecological limits to the invasion of Sargassum muticum in northern Spain. Aquat. Bot. 60, 251–263.

Araujo, R., Sousa-Pinto, I., Barbara, I., Quintino, V., 2006. Macroalgal communities of intertidal rock pools in the northwest coast of Portugal. Acta Oecologica-Int. J. Ecol. 30, 192–202.

Asif, M., Muneer, T., 2007. Energy supply, its demand and security issues for developed and emerging economies. Renew. Sustain. Energy Rev. 11, 1388–1413.

Astles, K.L., 1993. Patterns of abundance and distribution of species in intertidal rock pools. J. Marine Biol. Assoc. United Kingdom 73, 555–569.

Bergen, S.D., Bolton, S.M., 2001. L. Fridley, J. Design principles for ecological engineering. Ecol. Eng. 18, 201–210.

Bertness, M.D., Garrity, S.D., Levings, S.C., 1981. Predation pressure and gastropod foraging: a tropical-temperate comparison. Evolution 35, 995–1007.

Blockley, D., Chapman, M., 2006. Recruitment determines differences between assemblages on shaded or unshaded seawalls. Marine Ecol.-Prog. Ser. 327, 27.

Bolton, D., Mayer-Pinto, M., Clark, G.F., Dafforn, K.A., Brassil, W.A., Becker, A., Johnston, E.L., 2017. Coastal urban lighting has ecological consequences for multiple trophic levels under the sea. Sci. Total Environ. 576, 1–9.

Borenstein, M., Hedges, L.V., Higgins, J.P.T., Rothstein, H.R., 2009. Reporting the Results of a Meta-Analysis. Introduction to Meta-Analysis. John Wiley & Sons, Ltd.

Brittingham, M.C., Williams, L.M., 2000. Bat boxes as alternative roosts for displaced bat maternity colonies. Wildl. Soc. Bull. (1973–2006) 28, 197–207.

Brown, C.J., 2005. Epifaunal colonization of the Loch Linnhe artificial reef: influence of substratum on epifaunal assemblage structure. Biofouling 21, 73–85.

Browne, M., Chapman, M., 2014. Mitigating against the loss of species by adding artificial intertidal pools to existing seawalls. Mar. Ecol. Prog. Ser. 497, 119–129.

Bulleri, F., Chapman, M.G., 2010. The introduction of coastal infrastructure as a driver of change in marine environments. J. Appl. Ecol. 47, 26–35.

Burt, J., Bartholomew, A., Bauman, A., Saif, A., Sale, P.F., 2009. Coral recruitment and early benthic community development on several materials used in the construction of artificial reefs and breakwaters. J. Exp. Mar. Biol. Ecol. 373, 72–78.

Bussell, J.A., Lucas, I.A.N., Seed, R., 2007. Patterns in the invertebrate assemblage associated with Corallina officinalis in tide pools. J. Mar. Biol. Assoc. United Kingdom 87, 383–388.

Chapman, M.G., 2003. Paucity of mobile species on constructed seawalls: effects of urbanization on biodiversity. Mar. Ecol. Prog. Ser. 264, 21–29.

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

Chapman, M.G., Bulleri, F., 2003. Intertidal seawalls – new features of landscape in intertidal environments. Landscape Urban Plann. 62, 159–172.

Chapman, M.G., Underwood, A.J., 2011. Evaluation of ecological engineering of “armoured” shorelines to improve their value as habitat. J. Exp. Mar. Biol. Ecol. 400, 302–313.

Chapman, M.G., Underwood, A.J., Browne, M.A., 2017. An assessment of the current usage of ecological engineering and reconciliation ecology in managing alterations to habitats in urban estuaries. Ecol. Eng.

Connell, S., 2001. Urban structures as marine habitats: an experimental comparison of the composition and abundance of subtidal epibiota among pilings, pontoons and rocky reefs. Mar. Environ. Res. 52, 115–125.

Connor, E.F., McCoy, E.D., 1979. The statistics and biology of the species-area relationship. Am. Nat. 113, 791–833.

Cox, T.E., Baumgartner, E., Philippoff, J., Boyle, K.S., 2011. Spatial and vertical patterns in the tidepool fish assemblage on the island of O'ahu. Environ. Biol. Fishes 90, 329–342.

Cunha, E.A., Carvalho, R.A.A., Monteiro, C., Moraes, L.E.S., Araujo, M.E., 2008. Comparative analysis of tidepool fish species composition on tropical coastal rocky reefs at State of Ceara, Brazil. Iheringia Ser. Zool. 98, 379–390.

Dafforn, K.A., 2017. Eco-engineering and management strategies for marine infrastructure to reduce establishment and dispersal of non-indigenous species. Manage. Biol. Invasions 8, 153–161.

Dafforn, K.A., Glasby, T.M., Airoldi, L., Rivero, N.K., Mayer-Pinto, M., Johnston, E.L., 2015a. Marine urban sprawl: how can ecology inform the design of multifunctional artificial structures? Front. Ecol. Environ. 13, 82–90.

Dafforn, K.A., Glasby, T.M., Johnston, E.L., 2012. Comparing the invasibility of experimental “reefs” with field observations of natural reefs and artificial structures. PLoS One 7, e38124.

Dafforn, K.A., Kelaher, B.P., Simpson, S.L., Coleman, M.A., Hutchings, P.A., Clark, G.F., Knott, N.A., Doblin, M.A., Johnston, E.L., 2013. Polychaete richness and abundance enhanced in anthropogenically modified estuaries despite high concentrations of toxic contaminants. PLoS One 8, e77018.

Dafforn, K.A., Mayer-Pinto, M., Morris, R.L., Waltham, N.J., 2015b. Application of management tools to integrate ecological principles with the design of marine infrastructure. J. Environ. Manage. 158, 61–73.

Daryanavard, A., Abdoli, A., Ranjbar, M.S., Shokri, M.R., Kiabi, B.H., 2015. Spatial and temporal dynamics of the rock-pool ichthyofauna on Qeshm Island, Persian Gulf (Osteichthyes). Zool Middle East. 61, 133–138.

Davidson, T.M., Grupe, B.M., 2015. Habitat modification in tidepools by bioeroding sea urchins and implications for fine-scale community structure. Mar. Ecol. 36, 185–194.

Davis, J.L.D., 2000. Spatial and seasonal patterns of habitat partitioning in a guild of southern California tidepool fishes. Mar. Ecol. Prog. Ser. 196, 253–268.

Davis, J.L.D., 2001. Diel changes in habitat use by two tidepool fishes. Copeia 835–841.

Dumas, J.V., Witman, J.D., 1993. Predation by herring-gulls (larus-argentatus coues) on 2 rocky intertidal crab species carcinus-maenas (L) and cancer-irroratus say. J. Exp. Mar. Biol. Ecol. 169, 89–101.

Engel, C.R., Destombe, C., Valero, M., 2004. Mating system and gene flow in the red seaweed Gracilaria gracilis: effect of haploid-diploid life history and intertidal rocky shore landscape on finescale genetic structure. Heredity 92, 289–298.

Evans, A.J., Firth, L.B., Hawkins, S.J., Morris, E.S., Goudge, H., Moore, P.J., 2016. Drill-cored rock pools: an effective method of ecological enhancement on artificial structures. Mar. Freshw. Res. 67, 123–130.

Fairweather, P.G., 1988. Movements of intertidal whelks (Morula marginalba and Thais orbita) in relation to availability of prey and shelter. Mar. Biol. 100, 63–68.

Firth, L.B., Browne, K.A., Knights, A.M., Hawkins, S.J., Nash, R., 2016. Eco-engineered rock pools: a concrete solution to biodiversity loss and urban sprawl in the marine environment. Environ. Res. Lett. 11, 094015.

Firth, L.B., Schofield, M., White, F.J., Skov, M.W., Hawkins, S.J., 2014a. Biodiversity in intertidal rock pools: Informing engineering criteria for artificial habitat enhancement in the built environment. Mar. Environ. Res. 102, 122–130.

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

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

Garrity, S.D., 1984. Some adaptations of gastropods to physical stress on a tropical rocky shore. Ecology 65, 559–574.

Gibson, R.N., 1972. The vertical distribution and feeding relationships of intertidal fish on the atlantic coast of France. J. Anim. Ecol. 41, 189–207.

Glasby, T.M., 1999. Differences between subtidal epibiota on pier pilings and rocky reefs at marinas in Sydney, Australia. Estuarine Coast. Shelf Sci. 48, 281–290.

Glasby, T.M., 2000. Surface composition and orientation interact to affect subtidal epibiota. J. Exp. Mar. Biol. Ecol. 248, 177–190.

Goldingay, R.L., 2009. Characteristics of tree hollows used by Australian birds and bats. Wildlife Res. 36, 394–409.

Goss-Custard, S., Jones, J., Kitching, J.A., Norton, T.A., 1979. Tide pools of Carrigathorna and Barloge Creek. Philos. Trans. R. Soc. London B, Biol. Sci. 287, 1–44.

Gray, D.R., Hodgson, A.N., 1998. Foraging and homing behaviour in the high-shore, crevice-dwelling limpet Helcion pectunculus (Prosobranchia: Patellidae). Mar. Biol. 132, 283–294.

Green, J.M., 1971. Local distribution of oligocottus-maculosus girard and other tidepool cottids of west coast of Vancouver island, British Columbia. Can. J. Zool. 49 1111–&.

Griffiths, S.P., Davis, A.R., West, R.J., 2006. Role of habitat complexity in structuring temperate rockpool ichthyofaunas. Mar. Ecol. Prog. Ser. 313, 227–239.

Grimm, N.B., Faeth, S.H., Golubiewski, N.E., Redman, C.L., Wu, J., Bai, X., Briggs, J.M., 2008. Global change and the ecology of cities. Science 319, 756–760.

Gustavss, U., 1972. Proposal for a classification of marine rockpools on Swedish west coast. Bot. Mar. 15, 210–214.

Hawkins, S.J., Burcharth, H.F., Zanuttigh, B., Lamberti, A., 2010. Environmental Design Guidelines for Low Crested Coastal Structures. Elsevier.

Hendrickx, J.P., Creese, R.G., Gribben, P.E., 2015. Impacts of a non-native gastropod with a limited distribution; less conspicuous invaders matter too. Mar. Ecol. Prog. Ser. 537, 151–162.

Huggett, J., Griffiths, C.L., 1986. Some relationships between elevation, physicochemical variables and biota of intertidal rock pools. Mar. Ecol. Prog. Ser. 29, 189–197.

Johnson, M.P., 2001. Metapopulation dynamics of Tigriopus brevicornis (Harpacticoida) in intertidal rock pools. Mar. Ecol. Prog. Ser. 211, 215–224.

Johnston, E.L., Mayer-Pinto, M., Crowe, T.P., 2015. REVIEW: chemical contaminant effects on marine ecosystem functioning. J. Appl. Ecol. 52, 140–149.

Jordaan, A., Crocker, J., Chen, Y., 2011. Linkages among physical and biological properties in tidepools on the Maine Coast. Environ. Biol. Fishes 92, 13–23.

Jorger, K.M., Meyer, R., Wehrtmann, I.S., 2008. Species composition and vertical distribution of chitons (Mollusca: Polyplacophora) in a rocky intertidal zone of the Pacific coast of Costa Rica. J. Mar. Biol. Assoc. United Kingdom 88, 807–816.

Kita, T., Fukuyo, Y., Tokuda, H., Hirano, R., 1985. Life-history and ecology of goniodoma-pseudogoniaulax (Pyrrhophyta) in a rockpool. Bull. Mar. Sci. 37, 643–651.

Le Roux, D.S., Ikin, K., Lindenmayer, D.B., Bistricer, G., Manning, A.D., Gibbons, P., 2016. Effects of entrance size, tree size and landscape context on nest box occupancy: considerations for management and biodiversity offsets. For. Ecol. Manage. 366, 135–142.

Lintas, C., Seed, R., 1994. Spatial variation in the fauna associated with mytilus-edulis on a wave-exposed rocky shore. J. Molluscan Stud. 60, 165–174.

Macieira, R.M., Joyeux, J.C., 2011. Distribution patterns of tidepool fishes on a tropical flat reef. Fish. Bull. 109, 305–315.

Mahon, R., Mahon, S.D., 1994. Structure and resilience of a tidepool fish assemblage at Barbados. Environ. Biol. Fishes 41, 171–190.

Marsh, B., Crowe, T.M., Siegfried, W.R., 1978. Species richness and abundance of clinid fish (teleostei clinidae) in inter-tidal rock pools. Zoologica Africana. 13, 283–291.

Martins, G.M., Hawkins, S.J., Thompson, R.C., Jenkins, S.R., 2007. Community structure and functioning in intertidal rock pools: effects of pool size and shore height at different successional stages. Mar. Ecol. Prog. Ser. 329, 43–55.

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

Matias, M.G., 2013. Macrofaunal responses to structural complexity are mediated by environmental variability and surrounding habitats. Mar. Biol. 160, 493–502.

Mayer-Pinto, M., Johnston, E.L., Bugnot, A.B., Glasby, T.M., Airoldi, L., Mitchell, A., Dafforn, K.A., 2017. Building 'blue': an eco-engineering framework for foreshore developments. J. Environ. Manage. 189, 109–114.

Mayr, M., Berger, A., 1992. Territoriality and microhabitat selection in two intertidal New Zealand fish. J. Fish Biol. 40, 243–256.

Mering, E.D., Chambers, C.L., 2014. Thinking outside the box: a review of artificial roosts for bats. Wildl. Soc. Bull. 38, 741–751.

Metaxas, A., Hunt, H.L., Scheibling, R.E., 1994. Spatial and temporal variability of macrobenthic communities in tidepools on a rocky shore in Nova-Scotia, Canada. Mar. Ecol. Prog. Ser. 105, 89–103.

Mgaya, Y.D., 1992. Density and production of clinocottus-globiceps and oligocottus-maculosus (cottidae) in tidepools at Helby Island, British-Columbia. Mar. Ecol. Prog. Ser. 85, 219–225.

Morris, R.L., Golding, S., Dafforn, K.A., Coleman, R.A., 2017. Can coir increase native biodiversity and reduce colonisation of non-indigenous species in eco-engineered rock pools? Ecol. Eng.

Nakagawa, S., Santos, E.S.A., 2012. Methodological issues and advances in biological meta-analysis. Evol. Ecol. 26, 1253–1274.

Nakamura, R., 1976. Experimental assessment of factors influencing microhabitat selection by 2 tidepool fishes oligocottus-maculosus and oligocottus-snyderi. Mar. Biol. 37, 97–104.

Noble, D.W.A., Lagisz, M., O'Dea, R.E., Nakagawa, S., 2017. Nonindependence and sensitivity analyses in ecological and evolutionary meta-analyses. Mol. Ecol n/a–n/a.

Noel, L., Hawkins, S.J., Jenkins, S.R., Thompson, R.C., 2009. Grazing dynamics in intertidal rockpools: connectivity of microhabitats. J. Exp. Mar. Biol. Ecol. 370, 9–17.

Pandit, S.N., Cottenie, K., Enders, E.C., Kolasa, J., 2016. The role of local and regional processes on population synchrony along the gradients of habitat specialization. Ecosphere 7, 11.

Pinn, E.H., Mitchell, K., Corkill, J., 2005. The assemblages of groynes in relation to substratum age, aspect and microhabitat. Estuar. Coast. Shelf Sci. 62, 271–282.

Pulgar, J., Waldisperg, M., Galban-Malagon, C., Maturana, D., Pulgar, V.M., Aldana, M., 2017. UV radiation impacts body weight, oxygen consumption, and shelter selection in the intertidal vertebrate Girella laevifrons. Sci. Total Environ. 578, 317–322.

Richkus, W.A., 1981. Laboratory studies of intraspecific behavioral interactions and factors influencing tidepool selection of the wooly sculpin clinocottus-analis. California Fish Game 67, 187–195.

Sala, O., Chapin III, F., Armesto, J., Berlow, E., Bloomfield, J., Dirzo, R., Huber-Sanwald, E., Huenneke, L., Jackson, R., Kinzig, A., 2000. Global biodiversity scenarios for the year 2100. Science 287, 1770–1774.

Savard, J.-P.L., Falls, J.B., 1981. Influence of habitat structure on the nesting height of birds in urban areas. Can. J. Zool. 59, 924–932.

Scheibling, R.E., Melady, R.A., 2008. Effect of water movement and substratum type on vegetative recruitment of the invasive green alga Codium fragile ssp tomentosoides. Bot. Mar. 51, 341–349.

Schifman, L.A., Herrmann, D.L., Shuster, W.D., Ossola, A., Garmestani, A., Hopton, M.E., 2017. Situating green infrastructure in context: a framework for adaptive socio-hydrology in cities. Water Res. Res. n/a–n/a.

Schreider, M.J., Glasby, T.M., Underwood, A.J., 2003. Effects of height on the shore and complexity of habitat on abundances of amphipods on rocky shores in New South Wales, Australia. J. Exp. Mar. Biol. Ecol. 293, 57–71.

Sebens, K.P., 1991. Habitat structure and community dynamics in marine benthic systems. Habitat Struct. 211–234.

Segovia-Rivera, V., Valdivia, N., 2016. Independent effects of grazing and tide pool habitats on the early colonisation of an intertidal community on western Antarctic Peninsula. Revista Chilena De Historia Natural 89.

Thompson, R.C., Crowe, T.P., Hawkins, S.J., 2002. Rocky intertidal communities: past environmental changes, present status and predictions for the next 25 years. Environ. Conserv. 29, 168–191.

Threlfall, C.G., Mata, L., Mackie, J.A., Hahs, A.K., Stork, N.E., Williams, N.S.G., Livesley, S.J., 2017. Increasing biodiversity in urban green spaces through simple vegetation interventions. J. Appl. Ecol. 54, 1874–1883.

Underwood, A.J., Skilleter, G.A., 1996. Effects of patch-size on the structure of assemblages in rock pools. J. Exp. Mar. Biol. Ecol. 197, 63–90.

vanTamelen, P.G., 1996. Algal zonation in tidepools: experimental evaluation of the roles of physical disturbance, herbivory and competition. J. Exp. Mar. Biol. Ecol. 201, 197–231.

Williamson, C.J., Brodie, J., Goss, B., Yallop, M., Lee, S., Perkins, R., 2014. Corallina and Ellisolandia (Corallinales, Rhodophyta) photophysiology over daylight tidal emersion: interactions with irradiance, temperature and carbonate chemistry. Mar. Biol. 161, 2051–2068.

Wolfe, J.M., Harlin, M.M., 1988. Tidepools in Southern Rhode-Island, USA. 1. Distribution and seasonality of macroalgae. Botanica Marina 31, 525–536.

Zhuang, S.H., 2006. Species richness, biomass and diversity of macroalgal assemblages in tidepools of different sizes. Mar. Ecol. Prog. Ser. 309, 67–73.

Zuur, A.F., Ieno, E.N., Walker, N.J., Saveliev, A.A., Smith, G.M., 2009. Mixed Effects Models and Extensions in Ecology with R. Springer, New York.