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Can coir increase native biodiversity and reduce colonisation of non-indigenous species in eco-engineered rock pools?

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ABSTRACT

The expansion of built infrastructure in the marine environment threatens natural ecological communities at local and regional scales. An increasing interest in incorporating heterogeneity that is reflective of natural rocky shores into artificial structures through ecological engineering seeks to mitigate negative impacts. The addition of complex surfaces and novel habitats, such as water-retaining features, has been particularly successful at increasing biodiversity of marine infrastructures to date. Importantly, key habitat-forming groups, such as the complex turfing algae *Corallina officinalis* found on natural shores and their associated assemblages are still lacking from these eco-engineered features. Furthermore, whether observed biodiversity increases from eco-engineering are due to native or non-indigenous species remains largely unknown. Here, we investigated whether adding small-scale complexity (artificial turf) to artificial rock-pools ('flowerpots') on urban seawalls enhanced their effectiveness to increase native biodiversity. Responses of benthic invertebrates, algae, epifauna and fish in flowerpots with and without artificial turf (coir) were quantified. Contrary to existing literature, which reports an increase in biodiversity with an increase in complexity, no consistent effect of coir was seen on benthic, epifaunal or fish assemblages. Native species consistently occupied more than 95% of space in flowerpots while the proportion of non-indigenous species in flowerpots was small (<75% of the assemblage) regardless of treatment, and decreased over time. This result is promising, but warrants further investigation to determine if these trends reflect seasonal patterns or if non-indigenous species colonise early, but are replaced over time by native species. These are important considerations when planning large-scale deployments of eco-engineering features on seawalls to ensure that native species are targeted without increasing opportunities for non-indigenous species.

1. Introduction

A consequence of coastal urbanisation is the replacement of natural foreshores with human-made infrastructure (Dugan et al., 2011). In some areas of the world, more than 50% of the shoreline has been overbuilt by structures for a variety of purposes such as coastal defence (e.g. seawalls) and the provision of recreational (e.g. marinas) and/or commercial (e.g. aquaculture) facilities (Dugan et al., 2011). It has been well documented that the construction of marine infrastructure can negatively impact ecological communities at local and regional scales (reviewed by Dafforn et al., 2015a; Heery et al., 2017). For example, human-made structures generally support less diverse assemblages than natural habitats (e.g. Chapman, 2003), with greater numbers of non-indigenous species (Dafforn et al., 2009). At a regional scale, development can affect marine connectivity through facilitation and/or restriction of the movement of organisms and changes to trophic linkages (Bishop et al., 2017). The proliferation of marine infrastructure needs to be met with environmentally-sensitive management informed by solutions-focused research to sustain marine ecosystems and the valuable services they provide.

Increasing research efforts have tested the value and effectiveness of "ecological or eco-engineering" (as part of 'reconciliation ecology' in Rosenzweig, 2003) to create multifunctional marine infrastructure that benefits both humans and nature (Dafforn et al., 2015a). Ecological enhancement has been successfully achieved through the incorporation of complex surfaces (e.g. Chapman and

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Underwood, 2011; Coombes et al., 2015; Firth et al., 2014; Loke et al., 2014; Loke and Todd, 2016; Martins et al., 2010; Moreira et al., 2007; Moschella et al., 2005) and the creation of additional novel habitats (e.g. Browne and Chapman, 2011, 2014; Chapman and Blockley, 2009; Chapman and Underwood, 2011; Evans et al., 2015; Firth et al., 2014; Morris, 2016) in seawalls and other defence infrastructure to increase the overall heterogeneity of substrata. In many of the studies to date, adding a habitat to marine infrastructure attracted a different suite of species to the surrounding seawall, and thus increased the overall biodiversity (e.g. Browne and Chapman, 2014; Chapman and Blockley, 2009; Evans et al., 2015; Morris, 2016). In some cases, however, eco-engineering has been used to target an increase in specific species, for example those that are commercially valuable or endangered (Martins et al., 2010).

In Sydney Harbour, Australia the addition of water-retaining 'flowerpot' features to seawalls was particularly successful in providing a habitat for many species not found on the surrounding seawall (Browne and Chapman, 2014; Morris, 2016). Similarly, in the UK, shallow artificial pools drilled into breakwaters supported significantly greater species richness than the emergent rock (Evans et al., 2015). Whilst successful in increasing biodiversity, some species which were common in natural rock pools in the area did not colonise the artificial pools during the period of data collection (Browne and Chapman, 2014; Chapman and Blockley, 2009; Evans et al., 2015; Morris, 2016). For example, in flowerpots, the abundance of ephemeral turf algae was high, however, complex turfs like *Corallina* sp. did not colonise or had very low abundance, whereas they were common in natural rock pools (Morris, 2016). Further many grazing molluscs were absent from the pots (Morris, 2016). Equally, the main difference between natural and drill-cored rock pools in breakwaters was the absence of *Corallina* sp. in the latter (Evans et al., 2015). This may suggest that the artificial habitat created is unsuitable for certain species. Alternatively, the potential for colonisation may be limited if connectivity to natural populations is low (Bishop et al., 2017).

Coralline turfs provide a biogenic habitat for many mobile species, including those that are found as adults in natural rock pools and were not found in the flowerpots (Chapman et al., 2005; Morris, 2016). Plastic artificial turfs have been used effectively as mimics of natural coralline turfs with gastropod assemblages shown to be similar between artificial and natural habitats after two months (Kelaher, 2003a). With our increasing knowledge of the impacts of plastics in the marine environment, natural fibres are preferential to plastic turfs as they do not introduce microplastic contamination (Browne et al., 2007; Cole et al., 2011). Soft elements, such as Astroturf mats and coir, have been used in eco-engineering along river banks to enhance complexity and facilitate vegetation growth (Goodson et al., 2003; Hoggart and Francis, 2014; Vishnudas et al., 2006). Although the addition of soft materials have been used successfully in the enhancement of river walls, there has been limited testing of this approach for marine infrastructure (except see, Lavender et al., 2017), and no studies have tested a natural, as opposed to plastic, turf. In the absence of natural complex turfs colonising the flowerpots, we tested whether the addition of coir turf would enhance the effectiveness of the pots in increasing biodiversity.

The effect of the addition of coir turf to flowerpots may be twofold. As well as having a potential effect on the recruitment of mobile species, turfs occupy space, which is a limiting resource in marine hard substrate communities (Stachowicz et al., 2002). Thus, adding an artificial turf to flowerpots may reduce the availability of primary substrate necessary for the settlement of some native, but also non-indigenous, species (Sutherland, 1978). Artificial structures are considered hotspots for non-indigenous species (Bulleri and Airoldi, 2005; Dafforn et al., 2009; Glasby et al., 2007). Few studies have determined whether the species colonising eco-engineered habitats are native or non-indigenous in origin (except see, Sella and Perkol-Finkel, 2015). An eco-engineering initiative may be undesirable if the increase in biodiversity is due to non-indigenous species and investment in their large-scale application would not be appropriate if eco-engineered structures facilitate the establishment and spread of non-indigenous species. Of course, the success of any project depends on the specific management objective.

The objective of this study was to quantify the value of adding soft elements to hard eco-engineered structures to increase efficacy of native species enhancement. Benthic (macroscopic sessile and mobile organisms, >5 mm), mobile epifaunal (<5 mm, e.g. amphipods and isopods) and fish assemblages were measured. Specifically the hypotheses tested were: 1. Species density (defined as the number of species per sample) and abundance of sessile benthic organisms will be lower in flowerpots with coir compared to without coir; 2. There will be a greater abundance and species density of mobile benthic organisms in flowerpots with coir; 3. The abundance and species density of non-indigenous species will be lower in flowerpots with coir; 4. Species density and abundance of mobile epifauna will be greater in flowerpots with coir and 5. The relative abundance and species density of fish will be greater around pots with than without coir. Feeding behaviour of fish was also compared between flowerpots with coir, and it was further predicted that the number of bites (used as a proxy for feeding) will be greater at pots with coir.

2. Materials and methods

2.1. Study sites and materials

Concrete flowerpots (7L, 315 mm diameter, Fig. 1a), modified from those developed by Browne and Chapman (2011), were fixed to sandstone seawalls with a stainless steel bracket at mid-shore tidal level (1–1.3 m above chart datum) in two locations: Farm Cove (33.86°S 151.22°E) and Elizabeth Bay (33.87°S 151.23°E) in Sydney Harbour, Australia in January-February 2016 (Fig. 2). Mid-tidal height was used because of previous studies that have shown water-retaining features to be most effective for enhancing biodiversity at this height compared to low and high-tidal height (Browne and Chapman, 2011, 2014; Firth et al., 2013). Seawalls were characterised by a predominant cover of oysters (*Saccostrea glomerata*, Sydney rock oyster and *Crassostrea gigas*, Pacific oyster) at mid-tidal height, with non-encrusting (e.g. turf, *Ulva lactuca* and *Corallina officinalis*) and encrusting (*Hildenbrandia rubra* and *Ralfsia verrucosa*) algae present. Mobile animals present on the seawall included chitons (*Sypharochiton pelliserpentis*), starfish (*Parvulastra exigua*) and gastropods (e.g. *Cellana tramoserica*, *Montfortula rugosa*, *Siphonaria denticulata*, *Patelloida mimula* and *Morula marginalba*).

Two ~15 m sites at each location were chosen on the seawall, separated by ~30 m and 10 pots were deployed at each site; individual pots were approximately 1 m apart. These were submerged during high tide and retained water during low tide, which created 300 mm deep pools. In each site, five pots were randomly allocated to one of two treatments: 1) coir and 2) no coir. For the coir treatment, a coir panel (734 cm^2, 1.5 cm fibre length, approximately 168 fibres per cm^2) was fixed to the back wall of each flowerpot with adhesive (Selleys® 3 in 1 adhesive, sealant and gap filler, Fig. 1a).

2.2. Benthic assemblage

To determine if there was a difference between the benthic assemblage colonising flowerpots with and without coir, flowerpots were sampled 4, 8, 12, 16 and 32 weeks following installation. Cover of algae and sessile invertebrates were estimated using ten