Concrete ecological enhancement in the temperate marine environment

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Copyright Statement

An artificial rockpool at the Marineff project site in Poole Harbour, Dorset, UK. Image by Jessica Bone.

Abstract

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In the coming years, intertidal habitat loss is expected to increase as coastal development and sea level rise squeeze coastal ecosystems. Concrete coastal infrastructure is known to be a poor surrogate for natural intertidal reef habitats, but ecological enhancement can improve biodiversity and habitat availability on these structures. This thesis investigates one of the most commonly used interventions, the artificial rockpool, and how they affect biodiversity on urban coastlines in previously unquantified ways on the south coast of England. It will also examine how one of the most used engineering materials, concrete, may affect biological colonisation and the impacts colonisation may have on a concrete substrate in the marine environment. Firstly, the bioreceptivity of concrete in the coastal environment is reviewed, discussing the various methods by which bioreceptivity may be enhanced. The chemical bioreceptivity of concrete remains somewhat unclear, and emphasis should be placed on increasing the micro- and macro-scale superficial rugosity which is known to reliably increase colonisation. The impact of colonisation on concrete infrastructure is then reviewed, and it is suggested that bioprotective and biodeterioration of concrete substrates occur in tandem. Some species may offer more bioprotection than others, including calcareous encrusting species like oysters and barnacles. Finally, this section concludes with an experiment comparing the bioreceptivity of two low-carbon cement mixes with an ordinary Portland cement control. It is concluded after 1 year of subtidal immersion that bioreceptivity of the three mixes performs inconsistently between sites, with each mortar mix demonstrating some limited bioreceptivity. It is advised that local environmental conditions, seasonal timing of structure deployment and surface texture are prioritised when designing concrete-based ecological enhancements. The second section focuses on the use of retrofitted concrete artificial rockpools as ecological enhancements. Firstly, the behavioural interactions of fish and crabs with artificial rockpools and the adjacent seawall at high tide were monitored between April and October in a temperate harbour. Using video footage, it was possible to ascertain the length of time bass, shore crabs, and shanny spent in each habitat and what behaviours they engaged in. More time was generally spent sheltering within the seaweed-covered rockpools, whereas the barnacle-covered seawall also provided feeding opportunities. Conspecific behaviour, including pre-copulatory behaviour, was observed only in the rockpools. Further comparisons were made between the seawall and artificial rockpools at low tide in another urbanised harbour, at three tidal heights between mean tide level and high-water neaps. Although the middle and lower level rockpools contained more species, compared to the seawall the uppermost rockpools provided greater biodiversity benefits and elevated the presence of brown perennial seaweeds. Vertical arrays allow species to migrate up the tidal zone and find refugia in a warming climate. Finally, a vertical array of rockpools on sheet piling in another urbanised estuary retained a significant amount of sediment which was sampled to determine the presence of any infauna. This was compared to sediment cores taken from the local mudflats. Species typical of disturbed estuarine mudflats were identified in the retained rockpool mud, demonstrating an added benefit artificial rockpools can provide. As soft sediment habitats are often lost when coastal defences are constructed, sediment retaining interventions may provide some mitigation against this. It is evident that artificial rockpools can provide significant biodiversity benefits compared to an unenhanced seawall, through the provision of shelter for mobile fauna, provision of refugia at sensitive upper tidal levels and the retention of sediment for infaunal species. Through the use of artificial rockpools, combined with low-carbon, durable and rugose concretes, vertical intertidal infrastructure can be made to provide valuable habitat in a persistently threatened urban coastal environment.

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Integrated papers

In line with the alternative formats of thesis outlined within Bournemouth University's Research Degree Code of Practice, this thesis follows the integrated format. Details of the integrated papers and their publication status are listed below. For all co-authored publications, I confirm that I am the lead author and contributed over 75% of the substantive content of each paper as per the Code of Practice. Each published paper that has been included in this thesis has not been edited further since publication, except to reference sections where changes were made to formatting only. At the time of the final submission of the corrected thesis, the publication status of each chapter was the following:

Paper	Reference	Chapter	Publication Status
1	Bone, J.R., Stafford, R., Hall, A.E. and Herbert, R.H.J., 2022. The intrinsic primary bioreceptivity of concrete in the coastal environment – a review. *Developments in the Build Environment.*	Chapter 2	Published open access – available here DOI: 10.1016/j.dibe.2022.100078
2	Bone, J.R., Stafford, R., Hall, A.E. and Herbert, R.H.J., 2022. Biodeterioration and bioprotection of concrete assets in the coastal environment. *International Biodeterioration and Biodegradation.*	Chapter 3	Published open access – available here DOI: 10.1016/j.ibiod.2022.105507
3	Bone, J.R., Stafford, R., Hall, A.E. and Herbert, R.H.J., 2023. Inconsistent bioreceptivity of three mortar mixes in subtidal sites.	Chapter 4	Published open access – available here DOI: 10.1016/j.ecoleng.2024.107265
4	Bone, J.R., Stafford, R., Hall, A.E. and Herbert, R.H.J., 2023. Artificial rockpools create habitat refugia on seawalls at high tide.	Chapter 5	Under review, Ecological Engineering
5	Bone, J.R., Stafford, R., Hall, A.E. and Herbert, R.H.J., 2023. Vertical arrays of artificial rockpools on a seawall provide refugia across tidal levels for intertidal species in the UK.	Chapter 6	Under review, Science of the Total Environment
6	Bone, J.R., Stafford, R., Hall, A.E. and Herbert, R.H.J., 2022. Estuarine infauna within incidentally retained sediment in artificial rockpools. *Frontiers in Marine Science.*	Chapter 7	Published open access – available here DOI: 10.3389/fmars.2021.780720

Some of the results contained herein were presented at conferences and appeared on broadcast television shows:

- Marineff Conference in Caen, France in May 2022 – a 20-minute oral presentation of the findings of Chapter 7;
- Institute of Civil Engineers Coasts, Marine, Structures and Breakwaters Conference in Portsmouth, UK in April 2023 – a 5-minute oral presentation and A0 poster of early results of Chapter 5;
- ITV Meridian News in May 2023 – an as-live interview showcasing the rockpools included in Chapter 5;
- BBC Country File in December 2023 – a pre-recorded interview showcasing the rockpools included in Chapter 5.

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The greatest colleague a researcher could ever ask for.

Salem

2014-2021

Chapter 1. Introduction

1.1 Coastal development and ecological impacts

The terrestrial area adjacent to coastlines worldwide is associated with a growing human population via increased habitation, and development of socio-economic infrastructure. Coast-adjacent living has historically provided benefits to humans, such as facilitation of maritime trade and food provision, but was boosted by colonial interests in travel and trade and has since been augmented by the expansion of contemporary shipping trade (McGranahan et al. 2007). Additional benefits include the ecosystem and cultural services (Lakshmi 2021) that are intrinsically linked to natural coastal environments, such as nutrient cycling and water quality management (Airoldi and Beck 2007). In addition to trade and food provision, coastal areas are also required to sustain commercial and recreational activity. Some coastal towns and cities that have become tourist destinations have to cope with disproportionately high influxes of visitors and the infrastructure required facilitating them (Burke et al. 2001). For example, the annual summer Bournemouth Air Show on the south coast of England drew over 1 million visitors to the coastal town of Bournemouth over a single weekend in 2009, when the resident population as of 2011 was just under 200,000 people (Martin 2009; Bournemouth Borough Council 2018).

Historically, coastal settlements were established in naturally sheltered areas, but as the need for and use of coastal habitation has evolved over time, settlement has occurred in increasingly exposed areas that require modification. Additionally, for socio-economic growth to occur, significant modifications to coastal environs have been made. Coastal engineering, defined in this thesis as the anthropogenic modification of coastal environs, has been occurring for several millennia (Charlier et al. 2005). For example, ancient Egyptians utilised rock breakwaters to dissipate wave energy from harbour entrances as early as 2580 BCE (Tallet and Marouard 2014). “Modern” coastal engineering was considered to commence in the Napoleonic era (Franco and Verdesi 1999).

Coastal sea defences have been pivotal in the prevention and management of flood and storm damage risks to coastal infrastructure and assets. Frequently, sea defences must be paired with other, complementary, sea defences to counteract the negative effects of the initial structure. For example, erosional processes that would eventually undermine a seawall can be limited by the implementation of groynes to retain sediment at the toe (Williams et al. 2016; Anfuso et al. 2021). Subsequently, the interrelated interactions of sea defences can lead to a “domino” effect, necessitating their proliferation along the coast to manipulate physical and chemical forces intrinsic with coastal living.

Although artificial structures provide substrate to natural rocky shore species, artificial structures are not analogous to natural rocky shore biotic communities. Epibiotic assemblages and population size associated with artificial structures differ to natural rocky shores (Glasby 1999; Connell 2001; Chapman and Bulleri 2003; Bulleri et al. 2005) and in many incidences have lower biodiversity (Bacchiocchi and Airoldi 2003; Chapman and Bulleri 2003; Bulleri and Chapman 2004; Moschella et al. 2005; Bulleri and Chapman 2010; Chapman 2006; Underwood and Chapman 2006; Vaselli et al. 2008; Firth et al 2013; Hall et al. 2017) and a greater prevalence of non-native species (Mineur et al. 2012). In temperate shores of the northeast Atlantic, dominant species on artificial structures include the limpet *Patella vulgata*, fucoid algae, and barnacles such as *Austrominius modestus/Semibalanus balanoides*. For example, natural rocky shore habitats at Lyme Regis on the south coast of England, UK, had ten species more than local artificial structures (Pinn et al. 2005). Organisms particularly sensitive to desiccation stress, such as sponges, hydroids and ascidians, were occasional or absent on the artificial structures.

Assemblages on artificial coastal structures <2 years old are often dominated by ephemeral species, such as the green alga *Ulva* spp., and older structures (6-7 years) are often dominated by brown fucoids and greater abundances of Patellid limpets (Pinn et al. 2005).

Ecological enhancement is the provision of habitat in the built environment that improves the ecological condition of the site, over and above avoidance, mitigation and compensation (Arnold 2012). Eco-engineering "is the combination of engineering and ecological principles to reduce environmental impacts from built structures" (Dafforn et al. 2015; Chapman and Underwood 2011). It is, in effect, the facilitation of ecological enhancement by adding features within infrastructure that will increase habitat availability and biodiversity, either for a specific species or ecosystem, maximising ecological output (Bergen et al. 2001; Mitsch and Jorgensen 2003).

1.2 Artificial rockpools as ecological enhancement

Artificial rockpools have been used as ecological enhancement for several years, with Chapman and Blockley (2009) one of the first to demonstrate proof of concept by integrating recessed pools of water in a masonry seawall in Sydney Harbour, Australia. Since then, rockpool-type structures have been produced in rip-rap rock armour using a drill core to produce neat, cylindrical water-retaining pits (>30 mm diameter) (Chapman and Underwood 2011; Evans et al. 2016; Hall et al. 2018; Chee et al. 2020), or jackhammers (Ostale-Valriberas et al. 2018) and rock wheels (Hall and Bone 2023) to scrape away rock to produce a water-retaining depression. Rock armour may also possess naturally occurring depressions or recesses from the quarrying process that can be positioned on topside surface, a technique known as informed or passive positioning (MacArthur et al. 2020). Standalone pre-cast concrete blocks, many of which are now commercially available (Evans et al. 2019), can include a rockpool in the topside surface (Firth et al. 2014; Perkol-Finkel and Sella 2015; Waltham and Sheaves 2018) and these blocks can be integrated within a rip-rap structure or retrofitted around an existing structure. As demonstrated by Chapman and Blockley (2009), rockpools can also be integrated into vertical seawalls, either through inclusion of adapted bricks in masonry seawalls (Chapman and Underwood 2011; Heath and Moody 2013; Firth et al. 2014) or by using moulds and formliners to create recesses in concrete seawalls constructed *in situ* (MacArthur et al. 2019). However, of all artificial rockpool designs, retrofitted 'bolt-on' rockpools are one of the most common (Naylor et al. 2017) as they can be used with both existing and new vertical seawalls and attached to concrete (Morris et al. 2017; Hall et al. 2019; Drakard et al. 2023), steel (Bone et al. 2022), masonry (Browne and Chapman 2011; Browne and Chapman 2014; Morris et al. 2017; Morris et al. 2018) and wooden (Hall and Bone 2023) substrates. They are also usually smaller and therefore cheaper than standalone units, which means budget-limited research projects are more likely to use them.

Bolt-on rockpools have been demonstrated to enhance biodiversity compared to the adjacent seawall structure (Browne and Chapman 2014; Hall et al. 2019) and provide habitat for perennial habitat-forming seaweeds across a variety of environmental contexts (Drakard et al. 2023). In addition to providing an intertidal pool, they may also retain soft sediment (Waltham and Sheaves 2018) and may provide habitat for intertidal benthic fish at high tide (Morris et al. 2017). This thesis will investigate how bolt-on rockpools provide habitat for estuarine infauna through the accumulation of fine mud and how rockpools are used by mobile intertidal fauna at high tide compared to the adjacent seawall. In addition, this thesis will investigate how the vertical arrangement of artificial rockpools impacts biodiversity outcomes on a concrete seawall. The results will consolidate findings from early research conducted on the east Australian coast and contribute to growing evidence demonstrating the multitude of ways artificial rockpools can be used to increase biodiversity on coastal structures in both warm and cold temperate environments.

The majority of pre-cast rockpools are made using concrete, and many of the structures bolt-on rockpools are fixed to are also concrete (Bijen 1996; Kosmatka et al. 2008). Alongside the development of ecological enhancement solutions has been the development of the cementitious materials used to produce them. Much of this research has focussed on how to make concrete less hostile to marine life and in some cases actively encourage settlement of organisms through chemical cues. This proclivity for biological colonisation is known as bioreceptivity and, in tandem with the commercialisation of ecological enhancement products, some concrete formulas have been patented. This thesis will review the relationship between marine biological colonisation and concrete substrates, collating and discussing the existing literature, and reinforcing some of the findings through experimental work.

1.3 Outline of the thesis

The first paper (Chapter 2) reviews the bioreceptivity of concrete in the marine environment, drawing on a vast body of literature to determine if and how cementitious substrates can be improved for primary colonisation. The relationship between concrete and colonisation is further explored in the second paper (Chapter 3) by looking at how epilithic biota may impact cementitious substrates by providing bioprotective and biodeteriorative effects. The third paper (Chapter 4) closes the section focussing on concrete with a field experiment comparing the bioreceptivity of three different cement mixes deployed in Hamble Harbour and Poole Harbour, ensuring that some of the limitations of existing bioreceptivity experiments identified in Chapter 2 were not applicable in this study.

The second section focusses on the use of artificial rockpools as ecological enhancements in the intertidal zone of the south coast of England and begins with a paper (Chapter 5) that investigated the behaviour of benthic and demersal mobile fauna interacting with the concrete rockpools and adjacent seawall underwater at high tide in Poole Harbour. This comparison between seawall and rockpools elucidated how organisms use each habitat differently depending on their life history, demonstrating how both habitats may have ecological value. Further comparison between artificial rockpools and the adjacent seawall substrate is made in Chapter 6, comparing species abundance at three different tidal heights at low tide in Hamble Harbour, and how this influences biodiversity outcomes. In addition to the species abundance on the seawall and rockpools, analyses were also made of the biota colonising the small amounts of retained mud in the rockpools. The final paper (Chapter 7) investigated artificial rockpools that had exclusively retained mud and not water in Littlehampton Harbour, demonstrating how unexpected outcomes of ecological enhancements can still provide habitat.

1.4 Impact of the coronavirus pandemic on the research

In early 2020, the coronavirus pandemic disrupted life on a global scale, and research was also heavily affected. The installation of the artificial rockpools as part of the Marineff Project, originally due in March 2020, was delayed until July 2020 (Poole Harbour site) and October 2020 (Hamble Harbour site). Due to the uncertainty of restrictions in those early months of the pandemic, and the realisation that there were no useful reviews pertaining to the relationship between biological colonisation and concrete substrates, two literature reviews (Chapters 2 and 3) were produced in lieu of being able to collect experimental field data. This background knowledge then inspired the field experiment testing the bioreceptivity of three different mortar mixes in Hamble Harbour and Poole Harbour, which were deployed following the relaxation of lockdown restrictions.

The outcome is an interdisciplinary thesis that blends materials science and ecology and explores two related themes: cement and its use in eco-engineering and its relationship with intertidal biological colonisation; and cement-based artificial rockpools as habitats in the intertidal zone.

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Chapter 2. The intrinsic primary bioreceptivity of concrete in the coastal environment – a review

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2.1 Abstract

The proliferation of artificial coastal structures (ACSs) in the marine environment causes intertidal habitat loss and is a poor surrogate for natural rocky shores in terms of species richness, abundance, and community composition. As hard engineered coastlines increase, there is growing interest in how new concrete structures can facilitate improved habitat and biodiversity compared to existing concrete structures. Experiments that have substituted cement binder and aggregates in varying proportions and combinations have demonstrated that it is possible to enhance the primary bioreceptivity of concrete, either chemically or via microtopographical texture. This review synthesises key literature and identifies which concrete formulas prove most effective at enhancing bioreceptivity and those that have limited value, providing recommendations for coastal practitioners and for formulas that warrant further study. It is evident that the efficacy of chemical bioreceptivity of concrete is likely to be spatio-temporally limited (months) and enhancing surface roughness should be prioritised as a way to enhance colonisation. However, both chemical and physical methods require further investigation in within *in situ* marine settings for longer durations (>12 months).

2.2 Introduction

The construction of artificial coastal structures (ACSs, Figure 1) and hardening of coastlines worldwide is proliferating (Dugan et al. 2011), a phenomenon referred to as ‘ocean sprawl’ (Duarte et al. 2013; Duarte 2014), leading to the loss of intertidal habitat (Bugnot et al. 2021) and modification of sediment-based ecosystems (Bishop et al. 2017; Dugan et al. 2018). Concrete comprises a large proportion of coastal structures (Bijen 1996; Kosmatka et al. 2020) as concrete is considered a versatile, durable and cost-effective material (Alexander and Nganga 2016) for which there is currently no viable alternative in the marine environment (Scrivener 2014). However, when compared to other natural intertidal rock substrata, ecologically, concrete is considered an insufficient surrogate (Connell and Glasby 1999; Chapman 2003; Moschella et al. 2005; Vaselli et al. 2008; Pister 2009; Bulleri and Chapman 2010). As coastal development continues to threaten coastal ecosystems on a global scale (Bugnot et al. 2021), it is necessary to find ways to incorporate habitat within ACSs. Significant headway has been made in determining how this can be achieved with eco-engineering interventions, such as the introduction of water or mud retaining features, microhabitats such as cracks and crevices (Figure 2), and sloping and horizontal surface orientations (Strain et al. 2018; O’Shaughnessy et al. 2019).

Figure 1. Examples of artificial coastal structures (ACSs). Top left, clockwise; a concrete seawall; a stone masonry seawall; a rock groyne; a concrete dock. Images by Jessica Bone.

Figure 2. Examples of coastal eco-engineering. Top left, clockwise: An artificial rockpool containing water (Marineff Project); a Vertipool containing mud (Bone et al. 2022); a 'letter box' crevice in a concrete seawall; a cubic void containing water in a concrete seawall; a concrete seawall with a 'rockface' textured surface created with a formliner. Images by Jessica Bone.

There have also been many studies that have focused on improving the intrinsic primary bioreceptivity of concrete as the substrate material (references herein). Primary bioreceptivity is defined as the aptitude a material possesses for colonisation of biological life by virtue of the material composition and physical properties (Guillitte 1995) and, since the inception of this term, most research has focussed on lab-based studies, usually from a cultural heritage perspective with terrestrial conditions and biota (Sanmartín et al. 2021a). However, there are several studies that have looked to enhance the bioreceptivity of concrete in intertidal and subtidal settings, or to specifically attract marine organisms, by varying the binders (Perkol-Finkel and Sella 2014; Huang et al. 2016; McManus et al. 2018; Morin et al. 2018; Hayek et al. 2020; Natanzi et al. 2021; Ly et al. 2021), aggregates (Neo et al. 2009; Bedoya et al. 2014; Dennis et al. 2018; Hanlon et al. 2018; Ly et al. 2021; Potet et al. 2021) and additives used to modify its chemistry, pH (Guilbeau et al. 2003; Mos et al. 2019; Hayek et al. 2020; Hsiung et al. 2020), and surface porosity (Morin et al. 2018) and roughness (Pinheiro and Silva 2004; Neo et al. 2009; Sweat and Johnson 2013; Bedoya et al. 2014; Coombes et al. 2015; Dennis et al. 2018; Strain et al. 2018; MacArthur et al. 2019; Sedano et al. 2020). There may be some coastal settings or structures that are not appropriate for macroscale eco-engineering interventions, such as highly exposed shores, and so enhancing the bioreceptivity of the concrete material as a substrate for colonisation aims to maximise its ecological value in the absence of other habitat features. As the enhancement of ACSs gains traction in academic literature (Evans et al. 2022) and guidance (Naylor et al. 2017), in the coastal construction and engineering industry (Dale et al. 2011) and in legislation, it is necessary to clarify all options available to coastal practitioners to increase marine colonisation.

The purpose of this review is to synthesise and discuss how the bioreceptivity of concrete can be enhanced within the context of ACSs. It will focus on the superficial chemical and physical bioreceptivity of concrete, by virtue of its composition as per Guillitte's (1995) definition and stimulate conversation as to how coastal practitioners might enhance colonisation and biodiversity of concrete ACSs and underwater cultural heritage conservation by simply modifying the formula of concrete. Concrete formulas that prove to be of limited value or particularly effective will be identified and recommendations for further study will be made.

2.3 Scope of review

This review was conducted using online searches of terms pertaining to this field and databases including Web of Science, Scopus and Google Scholar was predominantly used to find relevant research, as well as suggested reading from these websites based on papers recently downloaded, and further references found therein. There are many terms synonymous with 'bioreceptivity', which were also searched to capture a broad range of published studies from a variety of journals and perspectives (ecological, engineering, materials science etc.). These synonymous terms included the following and, where applicable, were searched in both US and UK English and with hyphenated forms (i.e., 'bioreceptive' and 'bio-receptive'): bioreceptive, bioreceptivity, biological receptivity, biofouling, fouling, biocolonisation, biological colonisation. Studies examining bioreceptivity of concrete in the coastal zone in this review were predominantly conducted in the north-east Atlantic (50%), with the remaining studies being conducted in the north-west Atlantic, the north-west Pacific, the north Indian Ocean, the Mediterranean Sea, the Caribbean Sea and the Red Sea. No studies were conducted in the southern hemisphere, or on the continents of Africa, the Arctic or Antarctica.

It is important to note here the distinction between 'bioreceptive' concretes and 'eco-friendly/ environmentally friendly' concretes as the latter does not necessarily predispose the former. 'Eco-friendly' concretes usually involve the incorporation of materials that are either recycled, obtained from sources that minimise harm to the environment, obtained from local sources and so reduce the carbon footprint associated with shipping and transport, or are produced in an energy efficient manner (Wang et al. 2021). These benefits do not necessarily relate to or enhance bioreceptivity, but this can co-occur, for example the incorporation of recycled oyster shell to increase concrete surface texture (references herein). Use of 'abundance' in this review refers to the number of individuals of a given species, 'richness' refers to the number of species, and 'diversity' refers to the variation of organisms within a given dataset, though the latter two definitions are often used interchangeably.

There is a paucity of literature that focuses on the surface roughness intrinsic on the concrete surface as a consequence of its composition and mixing. Many studies focussing on roughness to enhance bioreceptivity often fail to define what 'roughness' constitutes in terms of scale, and roughness may co-occur with other features at the centimetre (cm) scale. It is often implied that roughness is very fine scale (micrometers to millimetres), but this is not always confirmed. Therefore, it is necessary to attempt to define the parameters of surface roughness (Strain et al. 2018) for the purposes of this review. According to Guillitte's (1995) definition, the intrinsic physical bioreceptivity of concrete occurs on a microscale where it's superficial porosity and micro-texture, or roughness, occur as a result of the material composition (Guillitte and Dreesen 1995). As no quantitative definition is used consistently within the eco-engineering field, though roughness can be measured quantitatively with Roughness Average (Ra), literature has been assessed using qualitative definitions. Enhanced concrete studies that focus on 'crevices', 'pits', 'cracks', 'rockpools', or variations thereof, have not been included, in addition to textures at the >cm scale or described as 'macro-scale'. Following Sanmartín et al.'s (2021a) review of Guillitte (1995), the expansion and clarification of what constitutes primary bioreceptivity in built structures allows for the inclusion of many studies that manipulate the topographic heterogeneity of ACSs with the addition of pits, grooves, holes, cracks and crevices. However, reviewing the literature that reports on macroscale features warrants their own paper and is both beyond the scope of this review, and already well summarised in Strain et al. (2018) and O'Shaughnessy et al. (2019). This review will focus predominantly on superficial porosity and roughness as a consequence of the concrete or mortar composition. However, it has been supplemented by some key studies (e.g., Tran et al. 2013; Coombes et al. 2015; MacArthur et al. 2019) that have manually modified the surface roughness of wet concrete, to provide further context about what textural benefits might be achieved if a concrete mixture was appropriately modified. Research on the surface heterogeneity of concrete to enhance bioreceptivity in the marine environment is scarce, and so studies focussing on terrestrial biota and rock material, such as limestone, sandstone and granite, have also been included (Table 1). Some studies have used 'surface texture' synonymously with roughness, and these studies are included where they meet the criteria stated above.

2.4 Chemical bioreceptivity

There is some evidence that concrete is already more bioreceptive when compared to other materials and leaches chemo-attractive cues that encourages the settlement of some species (Anderson and Underwood 1994). For example, Anderson (1996) conducted lab and field tests that determined that calcium hydroxide leachate from cement enhanced recruitment of Sydney rock oysters (*Saccostrea glomerata (commercialis)*). Several explanations as to the mechanism of this chemical cue were offered, including its indication of a potential site of high planktonic productivity and thus food resource, or its molecular role in triggering metamorphosis in larvae. Davis et al. (2017) found that, compared to High Density Polyethylene (HDPE) and granite, concrete was more bioreceptive to algal turf in mesocosm experiments, which may have been due to the dissolution of calcium from the concrete surface promoting growth. Dodds et al. (2022) found that compared to other artificial substrates, such as metal or plastic, concrete generally supports more species and in particular, more calcifying sessile invertebrates.

It is known that the chemical and mineralogical composition of a substrate can influence the preferential colonisation of some marine species (Bavestrello et al. 2000; Guidetti et al. 2004; Herbert and Hawkins 2006; Jones and Bennett 2017). Therefore, there is sufficient rationale for exploring how marine species might react to the chemistry of different concrete materials in the marine environment, by virtue of the choice of aggregate, binder and other additives.

2.4.1 Aggregate alternatives

An increasingly common experimental aggregate is shell (Richardson and Fuller 2013; Li et al. 2015; Nguyen et al. 2017; Varhen et al. 2017; Dahiru et al. 2018; Eziefula et al. 2018; Ruslan et al. 2021; Uddin et al. 2021; Han et al. 2022). In addition to utilising a waste by-product of the shellfish industry and potentially improving the affordability of concrete production, it is thought the use of crushed shell enhances surface roughness and provides chemical cues to induce settlement of mollusc larvae, and so enhance bioreceptivity of biogenic species such as oysters. Several taxonomic groups rely on conspecific chemical cues to settle in suitable habitat, such as barnacles (Browne and Zimmer 2001), sea urchins (Hay 2009) and oysters (Vasquez et al. 2013). Hanlon et al. (2018) replaced shale aggregate with ground non-native Pacific oyster (*Magallana gigas*) shell and compared the colonisation to a standard concrete mix without oyster shell. Following 6-months subtidal deployment in Falmouth, UK, it was revealed that the replacement of shale with oyster shell did lead to some differences in the initial community structure but did not recruit *M. gigas,* despite deployment coinciding with seasonal recruitment. However, molluscs on the concrete tiles containing oyster shell were >35% more abundant than on concrete tiles with shale. Despite this, the inclusion of oyster shell had a limited impact on species richness and abundance compared to other factors tested (surface complexity, surface orientation). Hanlon et al. (2018) noted that the high alkalinity of the fresh cement in the tiles may have negated the effect of shell chemical cues. Additionally, native molluscs may not have been able to detect chemical cues from a non-native oyster and so the use of non-native shellfish by-products may be redundant for native heterospecifics. Dennis et al. (2018) compared the bioreceptive performance of concretes made with different proportions of crushed whelk shell or hemp fibres to a concrete control with 10mm coarse aggregate. The experimental tiles were deployed on the coast of Wales, UK, for 12 months and percentage cover, species richness and biomass was determined. Initial biofilm colonisation measured *in situ* after approximately 2 months was greatest on hemp concrete but not significantly so. After 12 months, live cover was significantly greater on both shell and hemp concretes compared to the control concrete. Hemp concrete had higher species richness and significantly higher mobile species richness than both shell and control concrete. Despite hemp concrete demonstrating the greatest bioreceptivity, it was noted that further testing on its mechanical properties were required as it's generally deemed unsuitable for aquatic application (Dennis et al. 2018 and references therein). It was also noted that both hemp and shell concretes had noticeably rougher textures and it is therefore not possible to disentangle the influence of fine-scale surface roughness from the concrete chemistry as drivers of colonisation. Additionally, replications of this study should consider collecting time series data to validate that the results obtained in Dennis et al. (2018) and demonstrate when the biodiversity and assemblage structures of the different concrete mixes become divergent.

Crustose coralline algae are known to provide chemical cues for the settlement of coral (Morse and Morse 1984; Morse et al. 1988; Morse et al. 1996), starfish (Johnson and Sutton 1994) and abalone (Roberts et al. 2010). It was unknown if it was chemo-attractive to the fluted giant clam (*Tridacna squamosa*), so Neo et al. (2009) added crushed crustose coralline algae-covered coral rubble (CCACR) to concrete tiles in concentrations of 0%, 30% and 60% to determine settlement rates of fluted giant clam larvae. Settlement was greatest on the concrete samples containing 60% CCACR but the CCACR had no effect on the recruitment of juvenile clams 42 days after exposure to the tiles. It was postulated that the concrete was no longer leaching the chemo-attractant properties favoured by larvae during initial settlement or that the cues had become diluted beyond detection in the water. Alternatively, their habitat requirements may have changed post-settlement and CCACR cues were no longer prioritised. When variation in roughness was combined with CCACR concretes, rougher tiles were preferentially colonised by juvenile clams over smoother ones and the concentration of CCACR showed no significant effect.

Non-biological waste products have also been used as concrete aggregates, though the bioreceptive benefits of many examples are unclear. Bedoya et al. (2014) introduced crushed ceramics to concrete, but with no reference samples using typical aggregate to compare to, the value of adding crushed ceramics to concrete to enhance bioreceptivity remains unclear. Ly et al. (2021) used recycled glass from smashed car windows to produce concrete artificial reefs. Bioreceptivity experiments performed on mortar samples showed that the glass-based mortars were slightly more bioreceptive than other aggregates used (limestone sand and shell sand) at some sites. Glass-based concrete was used to produce the 3D-printed artificial reefs as it performed well in strength testing, and utilised recycled products.

It is evident that the mechanism by which mollusc shell may chemically enhance bioreceptivity is unclear. The evidence to support the exudation of chemoattractant from mollusc shell, particularly to target conspecifics, is limited and often confounded by other variables such as high alkalinity of the concrete or increased roughness due to the shell aggregate (Dennis et al. 2018; Hanlon et al. 2018; Potet et al. 2021). If chemical cues were leached, it is likely to be in insufficient concentrations due the amount of shell aggregate embedded within the concrete/ seawater interface. Any benefits appear to be temporally limited (Neo et al. 2009) or unpredictable. For example, only the aggregate in the surface is likely to exude cues but if the concrete surface is intensively weathered, more shell aggregate and thus chemical cue will be exposed. Burt et al. (2009) demonstrated the importance of secondary bioreceptivity (Guillitte 1995; Sanmartín et al. 2021a) following an experiment when coral recruitment on concrete substrate did not occur as expected. They emphasized that chemical cues of new substrates are short-lived, due to the colonisation of biofilms that mask the substrate surface and inhibit leaching, unless new surfaces are abraded (Cerrano et al. 1999). Therefore, the primary bioreceptivity of a substrate is likely temporally limited from when it is first placed in the environment, or freshly abraded, as the primary biological colonisation that occurs will limit contact between the substrate surface and successive organisms. Additionally, it is unknown if mixing shell with cement, which is chemically aggressive when wet, will lead to the decomposition of chemical cues rendering them inert. Thus far, it is the authors' opinion that the literature experimenting with the use of shell in concrete for its chemo-attractive properties is not conclusive enough to warrant their inclusion for this reason alone. However, there is evidence that shell concrete can lead to increases in abundance of certain taxa (Hanlon et al. 2018) and increase species richness (Dennis et al. 2018), either due to increased roughness (Potet et al. 2021) or accelerated carbonation. Georges et al. (2021) found that

CEMV based concrete with 20% shell aggregate underwent carbonation more rapidly than CEMV containing standard aggregate due to the additional voids caused by the shell aggregate. Shell aggregate reduces the carbon footprint of the concrete, by reducing the use of quarried aggregates, as well as using a waste product that would otherwise be discarded. Further study should identify how long the chemical cues of shell or coralline algae-based aggregates remain active *in situ* and if they are leached in any detectable quantities by target species in the field. Furthermore, surface roughness should be homogenised to eliminate texture as a confounding variable when determining chemical bioreceptivity.

2.4.2 Reducing pH and carbonation

Ordinary Portland cement (OPC) is the most common binder in concrete (Crow 2008; Kosmatka et al. 2008) and used in over half of ACSs (Lukens and Selberg, 2004; Perkol-Finkel and Sella 2014) but it is not considered the most bioreceptive. The lime content of OPC makes it alkaline with a pH of 12-14 (Taylor 1990; Manso et al. 2015), compared to a pH of around 8 for seawater, making it initially hostile to marine life, delaying the onset of colonisation (Grant 1982; Nandakumar et al. 2003). Atmospheric CO_2 reacts with $CaCO_3$ in the cement pore solution of concrete in an aging process known as carbonation with the speed of this process depending on the environment the concrete is exposed to (Hayek et al. 2020). This reduces superficial alkalinity to a pH of approximately 9-10 (Taylor 1990), by which time biological colonisation can begin in earnest (John 1988; Manso et al. 2015).

Manso et al. (2014a) attempted to lower the pH of OPC by adding boric and oxalic acid. The addition of the acid did not significantly impact pH and is therefore not likely to enhance bioreceptivity and, with respect to boric acid, could negatively impact mechanical properties making it unsuitable for use in industry. The exploration of alternatives to OPC includes magnesium-phosphate cement which is manufactured at a lower temperature than OPC and thus requires less energy to produce (Phair 2006). Unlike the high alkalinity of OPC, magnesium-phosphate cement (MPC) is neutral or slightly acidic, which allows concrete to be readily colonised, and demonstrates industry acceptable physico-mechanical properties (Manso et al. 2014a). Manso et al. (2014b) determined that MPC demonstrated a higher bioreceptivity for freshwater micro-algae (*Chlorella vulgaris*) compared to OPC under laboratory conditions after 10 weeks. Initial colonisation occurred more rapidly for MPC by one week and complete coverage was achieved 4 months sooner than OPC samples. Furthermore, Manso and Aguado (2016) determined that despite carbonated OPC specimens demonstrating up to 20% more voids and greater porosity than MPC specimens, the latter were more bioreceptive after 10 weeks, indicating that chemical composition of the binder played a greater role in colonisation than physical properties (porosity, surface roughness). However, Veeger et al. (2021) found the opposite in an experiment with MPC and OPC cements, with MPC concrete showing slower and less growth than OPC concrete after 8 weeks. Veeger et al. (2021) acknowledged these results may have been affected by the MPC blend used in their study, which contained unknown additives, and the choice of biofilm used in inoculation. It should be noted that these tests (Manso et al. 2014a; 2014b; Manso and Aguado 2016; Veeger et al. 2021) were conducted on concretes for use in the terrestrial environment, or using freshwater algae, and results may differ in marine environments.

The reduction of concrete pH has been explored for concrete in marine environments. Perkol-Finkel and Sella (2014) tested five concrete matrices in temperate and tropical marine settings that included lower proportions of OPC and additives to reduce pH and thus were designed to possess a lower alkalinity. In addition to meeting crucial industry standards for mechanical strength and durability, three of the concrete matrices were found to be more bioreceptive after 12 months, with greater live cover than concrete dominated by OPC. Additionally, surface roughness was also deemed to be of significant importance in enhancing bioreceptivity. In a lab-based study Hayek et al. (2020) compared the initial colonisation of bacterial biofilms on carbonated and non-carbonated concrete samples in seawater. Carbonated concrete had a lower surface pH and colonised immediately, whereas non-carbonated concrete experienced a lag of 7 days before initial colonisation occurred. Carbonated and non-carbonated CEMI and CEMIII mortar samples were submerged in seawater and their pH over time was compared. CEMI cement is pure OPC, whereas CEMIII is a mixture of OPC and 60% ground granulated blast-furnace slag (GGBS). The CEMI and CEMIII carbonated samples had initial pH values of 7.3 which remained constant over the immersion period, whereas the CEMI and CEMIII non-carbonated samples had initial pH values of 9.3 and 8.6 respectively and decreased to 7.5 and 7.3 respectively over the immersion period. Carbonated concrete initially performed better in terms of bacterial colonisation but after 60 days the bacterial biofilm on non-carbonated samples exceeded carbonated samples. CEMIII consistently outperformed CEMI and was recommended for use as marine concrete to enhance bioreceptivity.

Differences are also apparent in the colonisation of carbonated vs. non-carbonated concretes in algae and sessile invertebrates. Guilbeau et al. (2003) found pH neutral and carbonated concretes colonised more readily with algae after 16 days submerged on the coast of Florida, US, whereas non-carbonated samples were dominated by barnacles. It is worth noting that carbonation reduces the porosity of concrete by the precipitation of calcium carbonate within pore spaces (Dalod et al. 2014; Liu et al. 2018), which may reduce physical bioreceptivity (Tran et al. 2012). Despite this, larger pore spaces may remain unaffected by the carbonation process (Ngala and Page 1997), particularly in concretes containing GGBS (Gruyaert et al. 2013). However, this is likely to be ameliorated over time by abiotic and biotic erosive action exposing and expanding pore spaces (Tran et al. 2014).

Replacing the OPC content of concrete with an alternative binder that has a lower alkalinity has been found to enhance bioreceptivity (Manso et al. 2014a, Manso et al. 2014b; Perkol-Finkel and Sella 2014, Manso and Aguado 2016; Hayek et al. 2020) and carbonated concretes colonised more rapidly than non-carbonated concretes (Guilbeau et al. 2003; Hayek et al. 2020). Carbonation will naturally occur to concrete surfaces in contact with atmospheric carbon dioxide (Dooley et al. 1999, Hsiung et al. 2020) and so concrete-based ACSs will eventually increase in bioreceptivity as their pH decreases, and for the majority of their service life will be at an acceptable pH value for colonisation. It may be that a lag in bioreceptivity has temporal repercussions, depending on the time of deployment (Underwood and Anderson 1994; Nandakumar 1996) and the larval supply available when their pH has reduced, particularly if alkotolerant species, such as barnacles (Dooley et al. 1999, Guilbeau et al. 2003), dominate while surface alkalinity is still high. Hsiung et al. (2020) demonstrated that after 12 months there is little difference between the species richness, abundance and assemblage composition of carbonated and non-carbonated concrete tiles deployed in the UK and Singapore. Differences between the carbonated and non-carbonated surface porosity may have accounted for slight differences in species richness in Singapore, but this did not persist. Furthermore, Veeger et al. (2021) found that concrete samples, with their pH ranging between 11.6 and 12.2, showed significant biological growth with some samples becoming colonised within a week. Veeger et al. (2021) opined that the pH correlates with another determining factor, such as weathering enhancing surface roughness, but is not, itself, the factor responsible for colonisation. Mos et al. (2019) determined that initial colonisation of sea urchins (*Tripneustes gratilla*) on non-carbonated concretes showed greater survival and growth compared to greywacke (sandstone) and granite in warmed and acidified treatments. Thus, in a warming and acidifying ocean, the calcium carbonate leachate from fresh concrete may enhance species survival, though field testing is required to confirm these findings. Additionally, the value of this benefit may again be temporally limited.

2.4.3 Binder alternatives

GGBS and pulverised fly ash (PFA) are industry waste products that can be incorporated into concrete. GGBS is a by-product from iron ore extraction in a blast furnace. The molten iron is tapped off and all remaining materials form the slag, which is cooled and ground for use in construction materials. PFA is extracted from coal burning flues and can enhance durability of concrete and resistance to the deleterious effects of sulphate attack and chloride ion migration, which are common deteriorative issues in marine settings (Neville 2004; Santhanam and Otieno 2016). The use of pozzolans, broadly defined as finely ground siliceous and or aluminous materials, in concrete can reduce the surface pH (Guilbeau et al. 2003; Park and Tia 2004). GGBS and PFA can be used as a direct replacement for OPC up to 85% and 35% respectively (British Standards Institute 2011). However, pozzolan replacements can contain high concentrations of trace metals (Mullauer et al. 2015) which can leach out over long periods of time, although this is dependent on the microstructural characteristics of the cement paste (Jang et al. 2015). McManus et al. (2018) compared the bioreceptivity and leachate capacity of concrete mixes that substituted CEMI cement with direct replacement of 24% GGBS and PFA. Concrete tiles testing bioreceptivity were hung in Plymouth Sound, UK, for 7 weeks. Diatom coverage showed significant differences between the concrete mixes, but post hoc tests failed to reveal which concrete mix incurred greater coverage. The greatest native macro-fouling species richness was on the control treatment, which consisted of 100% OPC, and lowest on tiles containing GGBS. Assemblages on mixed tiles (24% PFA and 24% GGBS) were least similar whereas the control tiles showed the greatest similarity between assemblages. McManus et al. (2018) opined that that higher zinc content in GGBS limited colonisation due to its biocidal properties, but it was noted that OPC cement contained the highest concentration of copper, another biocidal metal, and yet performed best in the trial. Although there were significant results, it was unclear if heavy metal leachate significantly affects colonisation, particularly as the mixes with the lowest leachate concentration contained GGBS. There is no evidence that trace heavy metals transfer from hard substrate to epifaunal colonisation (Woodhead et al. 1985; 1986; Price et al. 1988). Additionally, many open systems are unlikely to retain sufficiently harmful concentrations of heavy metals (Foekema et al. 2016). It was also noted that far greater percentages of GGBS and PFA are used in industry as cement replacement, and that increasing the replacement from the 24% used in the study may enhance results.

Natanzi et al. (2021) compared the bioreceptivity of concrete mixes on the coast of County Meath, Ireland, with variation in the cement (100% CEMI vs. 50% CEMI and 50% GGBS), aggregate (limestone vs. granite) and presence of plasticizer (with or without). CEMI/ GGBS blends with granite aggregate demonstrated the lowest resistance to pH reduction compared to CEMI only with limestone aggregate samples. Samples were deployed on both sheltered and exposed sides of an intertidal breakwater for 1 month. CEMI/ GGBS blend tiles on the sheltered side had greater biomass of cyanobacteria, green algae and diatoms compared to CEMI only tiles. Conversely, on the exposed side, trends in microalgal biomass showed no significant difference. Morin et al. (2018) found that CEMIII concrete tiles outperformed CEMI concrete tiles in bioreceptivity of diatoms (*Entomoneis* sp.) after 28 days under laboratory conditions. Ly et al. (2021) compared the bioreceptivity of geopolymer binders with CEMIII cement. Geopolymer binder is made from alumina-silicates and alkaline reagents such as sodium or potassium hydroxide. CEMIII concrete blocks deployed subtidally in the UK, France, and Portugal had greater percentage cover than geopolymer concrete, though blocks deployed in Spain showed little difference. Biomass on geopolymer concrete was greater at 1- and 3-month intervals but overtaken at 6 months by CEMIII concrete. Additionally, CEMIII concrete outperformed geopolymer concretes in flexural and compressive strength. Hayek et al. (2021) found that CEMIII outperformed CEMI based concretes after 113 days immersion on the north French coast. Hayek et al.'s (2020; 2021), Natanzi et al.'s (2021), and Ly et al.'s (2021) studies demonstrate that higher proportions of GGBS in concrete can positively impact initial bioreceptivity, as well as better resisting chloride ion migration, thus mitigating deteriorative effects of seawater on concrete. Huang et al. (2016) found that a custom concrete mix ('green' artificial reef concrete) containing almost 70% GGBS in its cement formula performed to industry standard in mechanical testing and was readily colonised by *Ulva* spp. once deployed in the intertidal zone. The absence of alkotolerant barnacles and low levels of the hydration product portlandite suggested that the surface pH of the 'green' artificial reef concrete was lower than standard OPC. However, there was no comprehensive biotic data to support this conclusion, nor was the 'green' artificial reef concrete compared to an OPC or equivalent control. Subsequently, this study would benefit from replication involving OPC-based controls with time-series biotic data to confirm its findings. Hickling et al. (2022) compared the epibenthic colonisation of three concrete mixes (CEMII vs. CEMII and 10% addition of micro silica pozzolan vs. GGBS and alkali activation material) immersed subtidally on the Devon coast, UK, for one year. The 'CP' mixture (CEMII and 10% addition of micro silica) was rougher than the other mixes following washing of the surfaces following demoulding to expose the aggregate. Although no significant differences were found between species richness and diversity indices, percentage cover of sessile growth varied significantly (31-45%) and was greatest on the 'C' mixture (CEMII). Assemblage structure between the different mixes was also significantly different, likely due to the differing proportions of the bryozoan *Bugula* spp., which was greatest on the 'CP' mix. Although the exposed aggregate for the 'CP' was to represent the 'best practice' design, it makes it challenging to disentangle if differences between the mixtures were as a result of material composition or surface roughness.

The replacement of CEMI binder with CEMII or III, or the inclusion of GGBS with CEMI has been used to enhance bioreceptivity to diatoms, bacteria, cyanobacteria, green algae (Huang et al. 2016; Morin et al. 2018; Hayek et al. 2021; Natanzi et al. 2021) and overall percentage cover (Ly et al. 2021; Hickling et al. 2022). In order for GGBS to provide a clear positive impact for bioreceptivity however, it should replace OPC by a significant amount as recommended by McManus et al. (2018). Until further work can determine the threshold at which GGBS content becomes bioreceptively significant, it is recommended that GGBS replaces at least 50% of OPC as per Natanzi et al. (2021). Although lower alkalinity binders currently outperform OPC binders in the field, this may change as the ocean becomes more acidic. Davis et al. (2017) found that photosynthetic yield of algal turf on concrete in acidified treatments was 6% greater than other substrata (HDPE and granite). As many concrete ACSs being constructed now will aim to still be *in situ* in 50 to 100 years' time, further research should investigate the role different concrete matrices will play in bioreceptivity under warming and acidifying climate scenarios.

Veeger et al. (2021) found that adding bone ash (charred cattle bones) significantly increased bioreceptivity after 8 weeks of growth on OPC based concrete. Differences in algal growth between treatments containing no bone ash and bone ash were not apparent in earlier stages, which was thought to be because the phosphorus that the bone ash contained was not a limiting factor at that point in the experiment. Alternatively, the concrete may not have undergone sufficient weathering to expose sufficient concentrations of phosphorus-rich bone ash to the colonizing algae until several weeks had passed. The addition of phosphorus to a mineral substrate is known to promote biological growth (Jones and Bennett 2017) but can affect the durability of concrete, and so the inclusion of phosphorus-rich bone ash in concrete could be used to overcome this hurdle provided it does not exceed 20% binder replacement (Falade et al. 2012).

There are some examples of concrete that are ill-advised for use to enhance bioreceptivity, due to toxic or biocidal properties. For example, Dalod et al. (2014) compared the bioreceptivity of CEMI and calcium aluminate cement with green algae *Klebsormidium flaccidum* under lab conditions. It was found that calcium aluminate cement was not as prone to biofouling as CEMI, which is likely as a result of the high content of aluminate which has biocidal properties (Alexander and Fourie 2011; Herisson et al. 2013). Harilal et al. (2020) added nano-TiO_2 and nano-$CaCO_3$ to a conventional fly-ash based concrete and compared it with control concrete. Following six months immersion subtidally at Kalpakkam, India, the modified concrete was significantly less fouled than the control comparison due to the biocidal properties of the nano additives and their reaction products filling pores and reducing surface roughness. There are few examples where anti-fouling additives are added to a concrete mix, as it is more common practice for anti-fouling coatings to be applied superficially, which affects the quaternary bioreceptivity of concrete and is therefore beyond the scope of this review.

2.5 Physical bioreceptivity

The physical primary bioreceptivity of a material, and its influence on biofilm and meiofaunal colonisation, will affect secondary bioreceptivity by impacting the evolution of successive communities (Guillitte 1995). There is extensive literature documenting the biofouling capacity of building materials and it is generally agreed that surface roughness and porosity are key factors mediating biological colonisation (Deruelle 1991; Ortego-Calvo et al. 1995; Miller et al. 2006; Miller et al. 2009; D'Orazio et al. 2014), particularly small-bodied organisms (Strain et al. 2018). Additionally, roughness can mitigate against chemically toxic properties of a substrate. A biocidal coating applied to granite was found to increase bioreceptivity for algae and cyanobacteria under lab conditions, as the coating added micro-cracks to the surface, providing anchor points for colonisation and nullifying the impact of biocide (Sanmartín et al. 2021b).

2.5.1 Porosity

Increasing the porosity of concrete can enhance the deposition and accumulation of organic and inorganic particles, in addition to retaining moisture, which is known as extrinsic bioreceptivity. The deposition of spores and the use of pore spaces as habitat for biofilm taxa is referred to as intrinsic bioreceptivity (Guillitte 1995).

Guillitte and Dreesen's (1995) pioneering study determining the bioreceptivity of common construction materials (calcareous limestone, siliciclastic limestone, brick, mortar) examined the impact of porosity, roughness and mineral composition on the colonisation of terrestrial algae, cyanobacteria and moss. Aerated concrete was colonised rapidly, two weeks prior to the other materials. Vegetation coverage on the aerated concrete and siliciclastic limestone was 100% after six months, and both materials had a higher diversity of taxa than the mortar, brick and calcareous limestone. Initial colonisation patterns were correlated with material porosity, as a result of enhanced water retention and deposition of spores, with the most porous materials (aerated concrete and siliciclastic limestone) colonised rapidly with complete coverage and greatest taxonomic diversity after 6 months. Dubosc et al. (2001) demonstrated that more porous variations of OPC based concrete samples remained damp during the periods where the samples were not wetted, leading to a dominance of algal species less tolerant of desiccation on those concretes after 2 months. Although lab conditions were designed to emulate a terrestrial environment, these results could apply to the intertidal zone where concrete undergoes daily wetting and drying cycles. Giannantonio et al. (2009) found biofouling of terrestrial fungi was enhanced in concrete mixes that had higher water: cement ratios, which increased concrete porosity by reducing density after one week in lab conditions. This subsequently reduced the strength of the structure and so is not suitable for marine application. Giannantonio et al.'s (2009) findings were supported by Tran et al.'s (2014) field-based study that modified mortar porosity by using varying ratios of water: cement. The more porous samples demonstrated greater microbial fouling after a year and a half of exposure to natural fouling in a park in France. Snoeck et al. (2022) found that adding superabsorbent polymers to CEMI-based mortar samples to enhance the material's water retentive capability and macroporosity increased algae coverage after 10 weeks compared to controls.

Porosity may be less important than roughness in subtidal environments, where concrete is permanently immersed, and the retention of moisture is not a limiting factor for growth or survival of marine life. Tran et al. (2012) found porosity was not an important factor in determining bioreceptivity of mortars that were exposed to >80% humidity under lab conditions for 96 days. Instead, roughness was deemed to be the primary factor as it promoted adhesion of algal spores. In order to obtain a porous surface, Morin et al. (2018) used a vibrating table to create a dense layer at the concrete base and a porous layer at the top surface creating a bi-layered concrete. The porous layer comprised 20% of the total concrete thickness. Compared to a dense reference concrete, the porous bi-layered concrete had greater diatom growth after 28 days.

2.5.2 Roughness

The intrinsic bioreceptivity of different limestones were assessed by Miller et al. (2009) and both roughness and capillarity were important factors determining the amount of chlorophyll a measured on the sample surfaces. The rougher limestones were subject to greater microbial colonisation after 90 days and it was opined that the increased roughness reduced shear force stress and increased surface area available for colonising. By contrast, open porosity did not have as much influence on colonisation. With limestone, greater porosity may lead to greater evaporation of retained moisture, increasing environmental hostility and reduced colonisation.

When modifying the surface roughness of concrete materials to determine how roughness impacts bioreceptivity, the surface of the wet mortar or concrete is often poured against materials or formliners that will provide differing degrees of roughness when the cured concrete is struck (Pinheiro and Silva 2004). Alternatively, the test surface will be roughened with abrasive action either when wet or cured (Tran et al. 2012). Tran et al. (2013) found roughness created by abrading OPC mortar samples during setting with sponges enhanced colonisation of the algae *K. flaccidum* under lab conditions after 25 days. The surface textures were created with a ruler (smooth) and two sponges of differing texture (rough) and the mortars showed a linear relationship with bioreceptivity. The experiment was replicated in natural terrestrial conditions for up to 18 months where the effects of roughness were not as linear as when conducted under lab-based conditions (Tran et al. 2014). There was little difference between the bioreceptivity of the two rough treatments, but rough treatments outperformed smooth mortar. Coombes et al. (2015) compared the bioreceptivity of control, smooth, grooved and exposed aggregate OPC concrete treatments, which were achieved with standard casting and curing, wiping with a cloth, wiped with a coarse wire brush, and jet washed respectively. They were deployed intertidally on the southwest coast of the UK and monitored for cyprid settlement between May and November 2010 for six months. Intertidal barnacles preferentially settled on the grooved concrete, which corresponded to the size of the settling cyprids. The non-linear relationship of barnacle settlement with roughness was also observed in MacArthur et al. (2019), as barnacles preferentially settled on millimetre (mm) scale complexity. Recruitment of barnacles (*Chthalamus montagui* Southward, 1976) on natural limestones on the south coast of England, was determined by the number of potential settlement sites within pits and grooves on the rock surface, which was calculated from the length of cyprid larvae (Herbert and Hawkins 2006).

Bedoya et al. (2014) incorporated crushed waste ceramic as a replacement for typical aggregate in a Portland cement type 1 mix, and the sample plates with a higher percentage of large grain sizes had greater biodiversity after 8 months, although this was not significant. Dennis et al. (2018) found the intrinsic primary bioreceptivity of alternative aggregates (hemp and shell) may have also been due to the increased roughness of the unfinished tile surface after 12 months deployment intertidally. Neo et al. (2009) found that following six weeks of exposure to different concrete mixes, the clam *T. squamosa* preferred rough to smooth concrete tiles post-settlement, which they opined was due to the microscale topography providing some degree of shelter. Strain et al. (2018) found that rougher texture benefited subtidal sessile species and intertidal barnacles, branching coralline algae and, to a lesser extent, bivalves the most. Potet et al. (2021) found that after testing the bioreceptivity of nine different concrete formulas deployed subtidally on the French coast for 15 days, varying size class and substitution rate of oyster shell and siliceous sand aggregates, that micro-texture was more influential in the settlement of flat oysters (*Ostrea edulis*) than the formula chemistry. In the formula that substituted up to 50% of the aggregate with oyster shell, it was suggested that a microtopograpy approaching the same dimensions as settling larvae was the optimum rugosity for targeting flat oyster settlement.

Vivier et al. (2021) compared the photosynthetic health of marine biofilms growing on rough and smooth concretes versus plastic controls. After inoculation with natural marine biofilm under lab conditions for six days, microphytobenthic cells preferentially colonised negative reliefs in the rough concrete surface and had better photosynthetic performance and photoacclimation than biofilm on positive reliefs or smooth concrete. It was opined that negative reliefs shielded the biofilm from damaging levels of light and reduced hydrodynamic stress. Increasing open porosity and microscale roughness clearly also have a positive effect on algal percentage cover (Miller et al. 2009; Tran et al. 2012; Tran et al. 2013; Tran et al. 2014; Morin et al. 2018) though roughness may not have a linear relationship with the settlement of larvae, such as barnacles (Coombes et al. 2015; MacArthur et al. 2019). Therefore, enhanced roughness will not be suitable for the recruitment of all primary colonising taxa. Although conducted on acrylic panels, Sweat and Johnson (2013) determined that smoother surface textures favoured the adhesion of diatoms as a result of the increased contact between the material surface and the diatom cells. The cyprids of the barnacle *Amphibalanus* (*Balanus*) *improvisus* had greater preference for smoother surfaces than rougher surfaces (Berntsson et al. 1999; 2000). Cacabelos et al. (2016) found that rougher blocks of basalt seawall enhanced intertidal biodiversity over smoother blocks, but that barnacles preferentially settled on the smoother substrate. Sedano et al. (2020) found that microscale structural complexity was only significantly correlated with meiofauna, and larger scale complexity was more associated with sessile and vagile macrofauna. As noted by Sweat and Johnson (2013), recruitment to surface roughness may be dependent on the size of the settling organism, which is supported by Herbert and Hawkin's (2006) and Potet et al.'s (2021) observations with barnacle larvae and flat oyster larvae respectively. Subsequently it would be prudent to consider providing a variety of surface textures varying from smooth to coarse as increasing the number of habitats will enhance the biodiversity of the overall structure. Sedano et al. (2020) recommended pairing microscale roughness with macroscale habitats to boost sessile taxa. As with chemical bioreceptivity, most studies focusing on physical bioreceptivity are short term (<12 months) and do not demonstrate the effects of roughness and porosity on bioreceptivity long term.

2.5.3 Other factors influencing bioreceptivity

Hughes et al. (2013) postulated that the use of marine aggregates in marine concrete may enhance colonisation of algae due to the pre-existing algal spores and matter present in the aggregate and should be avoided to reduce fouling. However, most marine aggregates are sourced from relict geological or fossil deposits not associated with biological growth and are routinely washed for sorting (Highley et al. 2007). Additionally, wet cement is caustic (Peters 1984) and any remnants of biological content are unlikely to survive the aggressive chemical environment of setting and curing (Sanchez-Silva and Rosowsky 2008). Therefore, the reality of marine aggregates presenting a predisposition for biological colonisation, or 'fouling', is extremely unlikely, particularly when the practice of using beach sourced aggregate, as used in the concrete revetment Hughes et al. (2013) sampled, is relatively uncommon due to the issues associated with coastal erosion.

Colour can also affect bioreceptivity, although there is no literature to date for concrete substrates. Black and white acrylic tiles were submerged in the Sea of Oman for 20 days, with black tiles showing higher densities of fouling species compared to white tiles (Dobretsov et al. 2012). These findings of black versus white acrylic substrata were supported by Swain et al. (2007), who found higher settlement of green algae *Ulva* sp. and tubeworms on black surfaces after 14 days immersion on the Florida coast, and by Guenther et al. (2009) who found that the hydroid *Ectopleura larynx* preferentially settled on black tiles compared to other colours. Sanmartín et al. (2020) compared two granites that differed in colour due to the feldspar content (grey vs. red), but were otherwise similar in roughness, porosity and chemical composition. Significantly greater algal growth occurred on the red granite exposed to natural conditions in a UK woodland after 70 days, likely due to the reflection of red wavelengths that are known to stimulate algal growth more than green or blue wavelengths. It was noted that the effect of substrate colour would likely only impact primary succession as the substrate surface would soon be obscured with initial colonisation.

Table 1. Mechanisms affecting bioreceptivity of concrete in the coastal environment and potential application. The main findings of the different bioreceptive mechanisms are reviewed in this paper.

	Bioreceptive Mechanism	Main Conclusions	Advised Outcome	References
Chemical	Marine concrete	Calcium hydroxide leachate enhanced recruitment of Sydney rock oysters *Saccostrea glomerata* (Gould, 1850)	Further research required	Anderson 1996
	Shell aggregate - Oyster	Increased mollusc abundance Other factors tested (surface complexity/ orientation) showed greater impact	Use shell for textural benefits	Hanlon et al. 2018; Potet et al. 2021
	Shell aggregate - Crushed whelk Hemp aggregate	Hemp unsuitable for aquatic application. Shell and hemp increased surface roughness which was not possible to disentangle from chemical influence and so chemo-attractant benefits unclear	Use shell for textural benefits	Dennis et al. 2018
	Crushed crustose coralline algae-covered coral rubble (CCACR) aggregate	Settlement of target organism greatest on highest concentration of CCACR aggregate but no effect on recruitment after 42 days. When roughness was included in analysis the concentration of CCACR showed no significance.	Use for textural benefits	Neo et al. 2009
	Non-biological recycled aggregate	Crushed ceramics were not compared to reference samples so value unclear Recycled glass had limited to no bioreceptive value compared to limestone or shell sand	No clear bioreceptive benefits but beneficial for use in 'eco-friendly' concretes	Bedoya et al. 2014; Ly et al. 2021
	CEMI or ordinary Portland cement	High alkalinity when first cast leading to initial delays in colonisation	Substitute with bioreceptive binder where appropriate	Nandakumar et al. 2003
	Reducing the percentage content of ordinary Portland cement and replacing with lower pH alternatives	Concretes with lower proportions of ordinary Portland cement had greater live cover	Use for lower pH benefits though likely temporally limited	Perkol-Finkel and Sella 2014
	Carbonated concrete	Carbonated concrete colonised more rapidly and leads to an initial domination of algae. Also has a lower pH, though benefits are temporally limited, particularly in warming and acidifying oceans	Temporally limited value	Guilbeau et al. 2003; Mos et al. 2019; Hayek et al. 2020, 2021; Hsiung et al. 2020

	CEMIII or partial replacement of binder with ground granulated blast furnace slag	Percentage cover on CEM III was greater than CEM I concrete, though it was tested as a low percentage replacement for CEM I (24%)	Use for enhanced bioreceptivity at ≥50% GGBS content, though likely temporally limited	Huang et al. 2016; McManus et al. 2018; Morin et al. 2018; Hayek et al. 2020; Natanzi et al. 2021; Ly et al. 2021
	CEMII or partial replacement of binder with pulverised fly ash	CEM II was found not to enhance colonisation, though it was tested as a low percentage replacement for CEM I (24%)	Further research required	McManus et al. 2018
	Calcium aluminate cement, titanium dioxide additive	When compared to controls, calcium aluminate cement and concrete containing titanium dioxide were not as prone to biofouling due to biocidal properties of the metal content	Avoid use where possible	Dalod et al. 2014; Harilal et al. 2020
Physical	Increasing porosity	Retains moisture, organic and inorganic particulate matter, and larvae.	Use for enhanced bioreceptivity	Morin et al. 2018
	Increasing surface roughness	Greater surface roughness may increase surface area available for colonisation and facilitate attachment. Percentage cover is often greater on rougher textures, though this is not always the case.	Use a variety of surface textures for enhanced bioreceptivity	Pinheiro and Silva 2004; Neo et al. 2009; Sweat and Johnson 2013; Bedoya et al. 2014; Coombes et al. 2015; Dennis et al. 2018; Strain et al. 2018; MacArthur et al. 2019; Sedano et al. 2020; Hayek et al. 2021; Vivier et al. 2021.
	Colour	Black acrylic stimulated greater growth compared to white, but colour effects are likely to be temporally limited.	Further research required for coloured concrete	Swain et al. 2006; Dobretsov et al. 2013
	Pre-existing algal spores in marine sourced aggregates	Marine aggregates are predominantly sourced from relict geological or fossil deposits and thus do not contain biological material that would 'kick-start' colonisation. Biological material is unlikely to survive the caustic environment of wet concrete.	Unlikely to impact bioreceptivity	Hughes et al. 2013

2.6 Overview

Our review has shown that there are several methods by which the intrinsic primary bioreceptivity of concrete may be enhanced by virtue of its composition, and the pouring, curing and setting processes. The inclusion of crushed shell can enhance surface roughness and thus make concrete more bioreceptive, in addition to reducing its carbon footprint. However, its chemo-attractant value is likely to be spatially and temporally limited. Carbonating concrete also has limited long-term value but non-carbonated binders with a lower pH, such as CEMII or III, generally demonstrate greater bioreceptivity over OPC-based concrete provided the ratio of pozzolans is high enough. The use of GGBS and PFA is considered standard in marine concrete (British Standards Institute 2011) as OPC alone is insufficient at resisting chloride penetration (Smith 2016). Surface roughness and porosity generally enhance bioreceptivity and are likely more important than chemical bioreceptivity (Hayek et al. 2021), but it is important to include a variety of textures and porosity where possible, as marine organisms do not show a uniform response to increased roughness (MacArthur et al. 2019). Additionally, longer term studies are required to determine how physical bioreceptivity impacts successive communities, and how durable a rough and porous concrete surface is. Easily abraded concretes or concretes in highly exposed settings may become less rough and therefore less bioreceptive over time. The initial success of deploying bioreceptive concretes will also be reliant on extrinsic factors, such as larval supply, orientation, aspect, exposure, latitude (Strain et al. 2021), and disturbance and biotic interactions (Ferrario et al. 2016) will all play a role in longer-term success. The pH, and by extension bioreceptivity, of concrete may evolve in a warming/ acidifying ocean and this should be addressed with mesocosm experiments testing the pH and leachate of binders under climate change scenarios.

Enhancing bioreceptivity via the methods reviewed here is not a catch-all solution, as habitat preferences will vary among species and within life history stages. However, the magnitude of difference between the community structure and biodiversity of these patches will be site and context dependent (Becker et al. 2021) due to differences in local abiotic (salinity, pollution, climatic etc.) and biotic (larval supply, competition etc.) conditions. The majority, if not all, studies investigating bioreceptivity do not continue until a climax community has been reached, as noted by Dodds et al. (2022), which is likely due to the time constraints associated with this. Climax communities on ACSs could take between 5 to 20 years to form (Hawkins et al. 1983; Pinn et al. 2005; Coombes et al. 2011), and some authors have opined that complete succession may never occur in some contexts (Ferrario et al. 2016). As many ACSs are coastal defence structures, and therefore often situated on highly urbanised and exposed shores, there is an inherent local dynamism of a range of background abiotic and biotic disturbances that may render biotic communities in a constant state of flux. Enhancing the bioreceptivity of concrete may attenuate some of these stressors by increasing the likelihood of attachment and survival. However, there is a need for longer term studies investigating concrete bioreceptivity in the marine environment, particularly with appropriate references/ controls. Additionally, Sanmartín et al. (2021a) acknowledge there is also a succession to bioreceptivity, with primary bioreceptivity being superseded by the colonisation of biofilms and subsequent taxa. The deteriorative impacts of seawater, wetting drying cycles and biotic colonisation will produce secondary bioreceptivity over time, creating further micro and macroscale topographic complexity which will in turn modify biotic succession further still (Moschella et al. 2005; Firth et al. 2013b; Coombes 2014; Sanmartín et al. 2021a).

Such small-scale experiments are important for proof of concept to determine which concrete formulas and finishes provide optimal bioreceptivity, but it is crucial they are scaled up to structure scale to assess their performance as a larger contiguous habitat (Strain et al. 2018). Additionally, structure-scale pilots will provide crucial evidence that bioreceptive concretes can be successfully incorporated from design to delivery with negligible impacts on material sourcing, cost, construction and the structural integrity and service life of the structure. Prior to scaling up however, tile experiments may be a useful way to ascertain site-specific variation in available species and environmental parameters that will affect which physical interventions are best suited to the structure location. Post-construction monitoring would also be vital to determine success with appropriate control comparisons (Pioch et al. 2018). Small-scale trials achieve surface roughness either by leaving the surface 'unfinished' (Dennis et al. 2018) or by hand-finishing with sponges, brushes or rulers (Tran et al. 2013; 2014; Coombes et al. 2015), which are unlikely to be feasible or acceptable methods for structure-scale interventions. However, formliners for this purpose are now commercially available with a wide range of textures available, and concrete surfaces can be jet-washed to expose aggregates before fully set to enhance roughness. The recommendations for enhancing intrinsic primary bioreceptivity of concrete should be considered in tandem with macroscale interventions, such as water retaining and shading features in the intertidal, and holes and tunnels in the subtidal. Collaboration and communication between asset owners, coastal engineers and marine ecologists is essential (Naylor et al. 2012; Perkins et al. 2015) for the continued research into optimising concrete, that is a suitable compromise between maximising bioreceptive returns and material durability (Lubelli et al. 2021). This would also prevent the implementation of inappropriate features, such as unnecessary interventions in the upper shore zone, and gratuitous 'green-washing' (Firth et al. 2020). Regarding 'green-washing', caution should be exercised when referring to a concrete mix as 'bioreceptive', which implies an enhanced capacity for colonisation, when this has not been proven or simply has no observable detrimental impact to colonisation compared to more standard mixes (*sensu* Hickling et al. 2022). Further research should focus on replicating existing studies in different settings to identify which aspect of bioreceptivity is most beneficial in different contexts, which would allow for coastal managers to tailor bespoke bioreceptive concrete for their chosen site.

2.7 Conclusion

This review has demonstrated that there are several methods to enhance the chemical and physical bioreceptivity of concrete (Table 1). However, the evidence presented does not sufficiently justify the use of aggregates and binders to enhance chemical bioreceptivity, as exudation of chemical cues are likely to be spatio-temporally limited in open systems. Often the mechanisms of chemical bioreceptivity are unclear or conflicting, and sometimes rendered null and void following primary succession.

Most studies for both physical and chemical bioreceptivity report short term results (<12 months). The evidence for porosity and surface roughness suggests physical properties are more important in enhancing intrinsic primary bioreceptivity in concrete than chemical means, though the durability of rough surfaces in marine settings require further study. However, many of the aggregates and binders included in this review offer other benefits, such as the use of recycled materials and increased durability in the marine environment. Interdisciplinary coastal practitioners, including engineers and ecologists, should work together to create responsible, tailored bioreceptive solutions in ACSs that are suitable for the target site, target species and environment.

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Chapter 3 – A review of biodeterioration and bioprotection of concrete assets in the coastal environment

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3.1 Abstract

The deleterious effects (biodeterioration) and the protective benefits (bioprotection) of biological colonisation on manmade structures have long been debated. Lichens, biofilms, algae, bivalves and gastropods contribute both directly and indirectly to damaging substrata in the coastal zone which can enhance abiotic erosive forces that exploit biologically induced superficial damage. There is mounting evidence that these same species may also provide protective benefits. This debate often impacts approaches to managing fouling on concrete assets in the coastal environment. The net benefit or detriment a species or assemblage has on a structure is spatially and temporally dynamic and subject to the influence of various abiotic and biotic factors at different scales. However, the net outcome may be more pronounced under different contexts, particularly under warming and ocean acidifying climate change scenarios which is where further research should focus. Additionally, as bioprotection represents a potentially valuable ecosystem service, it supports the argument for enhancing habitat availability and biodiversity on artificial coastal structures via ecological enhancement. Quantifying bioprotection in useful metrics, such as monetary value or time added to serviceable life, would help demonstrate the benefits of bioprotective species in a meaningful way.

3.2 Introduction

The biological colonisation of artificial coastal structures is subject to a conflict depending on the perspective. Either biological colonisation is to be discouraged and avoided (biofouling) to prevent deleterious effects on the concrete substrate (biodeterioration, bioerosion, biocorrosion etc.) (Lebret et al. 2009; Hughes et al. 2013a; Hughes et al. 2013b), or it is to be facilitated to mitigate and/or compensate for habitat loss or create biodiversity gain and the ecosystem services associated with potential protective effects (bioprotection). Often, the porosity and surface roughness that asset managers and engineers cite as the main drivers for biofouling and seek to ameliorate (Harilal et al. 2020), eco-engineers and ecologists wish to enhance to promote biological colonisation (Guilitte 1995; Coombes et al. 2015).

Much research has focussed on either discouraging or promoting biological colonisation of concrete structures but often does not consider the potential benefits of the other. Reduced porosity in concrete often equates to denser and stronger concrete (Claisse et al. 2001; Neville 2011; Singh et al. 2018; Othman et al. 2021) which also makes it more challenging for biofilms and epibiota to colonise. Colonising biota are known to have potentially deleterious effects due to boring activity, the secretion of organic and inorganic acids, and penetration of attachment structures such as rhizomes and byssal threads (see Biodeteriorative Effects). When designing bioreceptive structures, greater porosity is recommended to promote colonisation of biofilms (Guillitte and Dreesen 1995; Morin et al. 2018) which facilitate successive organisms, such as macrophytes and invertebrates (Dubosc et al. 2001). Absent from much of the latter research is consideration for the biodeteriorative effects on concrete structures, how this may affect service life and thus the associated ongoing maintenance costs and labour. Subsequently, it is not yet fully accepted in industry how biological colonisation might offer protective benefits, such as moderating thermal and humidity regimes and wetting/ drying cycles

(Coombes et al. 2013; 2017), as well as buffering weather-induced stressors such as wind and wave action (Gowell et al. 2015).

The long-term goal of ecological engineering research is to incorporate artificial habitat features into coastal infrastructure from the design and planning stage, as opposed to the more commonly occurring retrofit, but first the concerns of asset owners must be addressed before acceptance and integration can occur on an industry-wide scale. The biodeteriorative and bioprotective effects of biological colonisation on intertidal hard substrates will be reviewed and the conflict will be addressed and discussed, with a focus on concrete materials. Finally, suggestions for further research will be made.

3.3 Scope of the paper

Online searches were performed using websites Web of Science and Google Scholar, using terms that would return relevant research and the references therein. 'Bioprotection' and 'biodeterioration' are synonymous with many other similar terms depending on the field of expertise and author perspective. These terms, where applicable, were searched in both UK and US English, hyphenated forms (i.e., 'bioprotection' and 'bio-protection') and different tenses (i.e., 'bioprotected', 'bioprotective'). For literature on 'bioprotection', the following terms were used to search: biostabilisation, biological protection, biological stabilisation, bioconstruction. For 'biodeterioration', the following were used to search: bioerosion, biological erosion, biodegradation, biological degradation, biogenic dissolution, biological decomposition. 'Biofouling' and 'fouling' are terms used to describe the undesired colonisation of built structures and were also used to search for literature on the impacts organisms have on hard coastal substrates. 'Biodeterioration' in this review is used as a proxy for an agglomeration of synonymic words including but not limited to: bioerosion, biodegradation, biogenic dissolution, biological decomposition/ erosion and the processes and mechanisms associated with these terms (*sensu* Davidson et al. 2018). Similarly, 'bioprotection' will be used as a proxy for biostabilisation and bioconstruction where relevant and includes active (direct removal of material) and passive (facilitation/ acceleration of other weathering and erosive action) mechanisms (Naylor et al. 2002; Naylor et al. 2012).

This review will focus on concrete structures in the intertidal zone but will draw on research looking at the biodeterioration and bioprotection of rock substrates (natural or heritage), and in terrestrial and freshwater environments to provide additional context where the equivalent research in a marine setting is lacking. It includes a mixture of lab-based and field-based experiments and observations from a range of academic peer-reviewed journals. 'Concrete' here refers to a composite material formed by mixing cement, both coarse and fine aggregate and water (British Standards Institute 2013) and may contain further admixtures. 'Mortar' is similar to this but does not include coarse aggregate. Extrapolating impacts of biological colonisation on rock substrates to concrete should be interpreted cautiously. However, given that concrete often contains, in varying quantities and grades, rock aggregates, there can be a fair assumption that it may also be vulnerable to the same biodeteriorative forces, particularly in a chemically aggressive intertidal environment.

The coastal environments included in this review focus is the intertidal environment between Extreme Low-Water Springs and Extreme High-Water Springs but can include the splash zone. It includes coasts of varying aspect to sunshine and exposure to wave action. It also includes fully saline habitats, and estuaries where the salinity would be significantly reduced and highly variable, depending on volume of water discharge from the rivers. Most papers are from temperate environments, or simulate temperate conditions, but a limited number of tropical examples also feature. Nearly all are conducted in field sites in the Northern Hemisphere, and predominantly in the North Atlantic. Concrete in the marine environment is vulnerable to a wide variety of deteriorative forces which are well summarised in Santhanam and Otieno (2016), with concrete in the intertidal and splash zone (the region above extreme high water spring tides) often considered the worst exposure categories. The ingress of chloride ions from seawater salts leads to the corrosion of steel embedded within reinforced concrete, which can result in rapid surface deterioration through spalling, where a section of concrete delaminates from the substrate. The chemical attack from chloride and sulphates can alter the microstructure of the cementitious composites (Neville 2004), which can increase the risk of cracking and loss of strength. Chemical attack reduces the integrity of the concrete at its surface, increasing its vulnerability to salt weathering, where salt from the sea is deposited following a period of drying in intertidal conditions, causing expansive pressure and damage. The present paper acknowledges that bioerosion and other physical and chemical forms of weathering often occur in tandem (Coombes 2014), particularly in the coastal environment where organisms may facilitate weathering and erosion through their activity or removal (Naylor et al. 2002; Naylor et al. 2012), but these interacting mechanisms are not explored in depth and the focus remains on direct biological deterioration.

3.4 Biodeteriorative effects

Organisms can facilitate physico-chemical weathering and erosion but their impacts on hard substrates can also promote the colonisation or behaviour of other organisms, as explained by Coombes (2014), and the dominance of a given taxonomic group or morphology will depend on tidal height (Trudgill 1987; Simms 1990). For example, weakened substrate surfaces from boring organisms can enhance the removal of particulate material by grazers (Schneider and Torunski 1983). The implications for concrete intertidal infrastructure are dependent on the lithologies of the aggregates and the chemical composition of cement used.

3.4.1 Microorganisms

Microorganisms, such as bacteria, fungi, micro-algae, and composite organisms, such as lichen, have bioerosive effects on both soft and hard rock and marine-grade concrete (Krumbein 1988; Morton and Surman 1994). These form ‘biofilms’, a mucilaginous matrix of extra-cellular polymeric substances (EPS) containing bacteria, protozoa, and diatoms (Wetherbee et al. 1998; Decho 2000). The metabolic activities of the organisms within this matrix play a role in the solubilisation of metal ions and the decomposition of substrate materials (Eckhardt 1985; Flemming 1993; Morton and Surman 1994; Cwalina 2008; Scheerer et al. 2009). Euendoliths (boring microorganisms) can be exceptionally abundant on limestone coastal rock, with up to half a million euendolithic filaments present in a single square centimetre (Schneider and Le Campion-Alsumard 1999). Biophysical weathering by biofilms can occur through the expansion and contraction of cells through wetting and drying cycles (Moses and Smith 1993; Gomez-Pujol et al. 2007) and the extension of hyphae (branching filaments) and growth into faults between rock crystals, and cement paste and aggregate. This leads to the creation of fissures and cracks, which are further weathered by mechanical erosion.

Coombes et al. (2011) studied the biophysical erosion of Cornish granite, Portland limestone and marine-grade concrete deployed in the intertidal in Cornwall, UK. EPS growth was present on all materials, but the spatial pattern of colonisation and erosive mechanisms differed between them. Microscopic boreholes were superabundant on the limestone and abundant on the concrete but almost absent on granite. Where boreholes were particularly dense, they coalesced, leading to collapse and loss of surface material, subsequently producing a fine-scale surface roughness not present at the start of the study. EPS thickness on granite was greatest when microtopographical features, such as mineral grain boundaries and ridges, were present. Owing to granite's hardness, endolithic growth was not recorded in the samples but was present in limestone and concrete. Biological crusts were superabundant on the concrete as a result of the chemical reaction between seawater salts and cement paste, leading to precipitates such as gypsum and brucite. The leaching of these precipitates ultimately leads to minor material loss, with brucite exhibiting dichotomous characteristics; expansion within the cement paste, increasing risk of cracking, but also sealing pores as an insoluble precipitate and thus preventing further leaching and seawater penetration (Costa and Appleton 1999; Neville 2004). Concretes that have greater porosity may exhibit greater EPS growth and penetration than dense concretes due to increased moisture retention and the ability for organisms to adhere to inner surfaces, and thus favourable growth conditions (Tamai et al. 1992; Ohshima et al. 1999; Dubosc et al. 2001; Vivier et al. 2021). This enhanced porosity, for example in CEMV concrete, can allow greater penetration of biodeteriorative agents (Georges et al. 2021). The lithological microorganism colonisation on these materials dictates in the long term their surface geomorphology as well as the subsequent ecological succession and the indirect bioerosive effects this will incur. The desiccation of biofilms themselves may also result in the loss of surface material, as the contraction of the EPS removes mineral grains from the substrate (Guillitte and Dreesen 1995). Coombes et al. (2011) did not deem EPS colonisation to pose a significant risk to the durability of a coastal defence structure over its service life.

3.4.2 Algae

Macrophytic algae are ubiquitously regarded as fouling organisms due to the increased drag (Fletcher 1988, Yebra et al. 2004, Schultz 2007), loading and fatigue damage (Edyvean et al. 1988; Yan et al. 2006), blocking of pipes, and slip hazard for the public (Lebret et al. 2009). They also contribute to bioerosion of coastal structures. The biophysical weathering that occurs as a result of algae colonisation is primarily via the penetration of attachment structures such as hyphae and holdfasts (Morrison et al. 2009). The penetration of the holdfasts of the temperate brown seaweeds (*Ascophyllum nodosum* and *Fucus vesiculosus*) of up to 1.5 mm and 4 mm respectively has been observed in Galway granite and Carboniferous limestone, with the crustose algae *Lithothamnion* sp. also demonstrating changes to rock surfaces (Morrison et al. 2009). Structured light microscopy (SLIM) and scanning electron microscopy (SEM) showed both *A. nodosum* and *F. vesiculosus* holdfasts exploited microfractures within the rock, such as intercrystalline boundaries and cleavage planes, and prized minerals apart as well as engulfing disaggregated fragments into algae tissue. It was anticipated that the intermittent wetting/ drying of the intertidal alga would enhance their bioerosive capacity on a micro-scale due to the expansion and contraction of alga tissue (Figure 3). This phenomenon was also observed by Hughes et al. (2013a) who found microscopic *Ulva* sp. filaments penetrating the cement paste and adhering to exposed fine aggregate particles of a concrete revetment. As with rock, it is evident that algae attachment structures can exploit weaknesses between cement paste and aggregate interface. As observed by Coombes et al. 2011 with EPS growth exploiting microtopographical features on granite, Hughes et al. (2013b) demonstrated with SEM micrographs algal filaments behaving in a similar manner on degraded concrete surfaces with exposed aggregate fibres. Where cement paste had eroded, leaving aggregates exposed, algae filaments were better able to adhere to and penetrate the concrete surface.

Large brown algae species, such as bull-kelp (*Durvillaea antarctica*), may contribute to coastal erosion via a phenomenon known as 'kelp plucking'. Following storm activity, Smith and Bayliss-Smith (1998) found that dislodged kelp removed rock attached to their holdfasts, contributing to intertidal downwearing of rock platforms. The force of removal may also introduce faults in the local rock, increasing the area vulnerable to further weathering.

Jayakumar and Saravanane (2009) identified that concrete subjected to epiphytic growth of the subtropical macroalgae *Chaetomorpha antennina* lead to the dissolution of calcium within the concrete and alteration of the surface material on the Indian coast. They demonstrated that *C. antennina* contained organic acids, but comparisons of the concrete condition were made between concrete samples in potable water in the lab, colonised concrete samples in seawater in the lab, and colonised concrete samples from the intertidal zone. Subsequently, without an uncolonised concrete sample in seawater to compare results to, the relationship between how the combination of *C. antennina* and seawater affect the concrete is unclear. Further work by Jayakumar and Saravanane (2010) and Jayakumar et al. (2011) replicate the study with *Ulva fasciata.* In one study (Jayakumar et al. 2011) control concrete in potable water is analysed, and in another (Jayakumar and Saravanane 2010) the control concrete is kept in saline water. The energy-dispersive X-ray spectroscopy (EDAX) graphs show remarkable similarities between the mineralogy of the colonised concrete samples in seawater and the control concrete in saline water. Comparing the EDAX graphs of the potable water and saline controls from the 2010 and 2011 papers, it is possible to see that the mineralogy of the saline water concrete and potable water concrete differ. This suggests that saline/ seawater also plays a role in the dissolution of calcium from the concrete in these studies, which is known to occur (Buenfeld 1984). It is also unknown if the organic acids identified in the alga are present in a chemically significant concentration and how these acids come into contact with the concrete (e.g., diffused into the water or leached into concrete via holdfast tissue). Additionally, the studies ignore the impact of biophysical weathering by algal attachment structures which can enhance abiotic chemical weathering (Griffin et al. 1991). Welton et al. (2003) demonstrated that when calcium rich stone was immersed in autoclaved tap water, calcium was released into the liquid. When microalgae were present, calcium was absorbed from the liquid indicating that microalgae utilise calcium from the substrate but incorporate it indirectly via its leaching in the presence of water. This is emphasised in Guillitte and Dreesen's (1995) study which examined the bioreceptivity of common building materials in lab conditions. Upon using a nutrient-rich solution to enhance colonisation, they noted that the polystyrene rests in which the building materials were held were also colonised by vegetation, suggesting that colonisation was primarily dependent on exogenous nutrients and not the inherent nutrient content of the material.

The decay of marine algae causes the release of hydrogen sulphide and dimethyl sulphide (Keller 1989) with concentrations of up 600 ppm recorded in decomposing seaweed in seawater (Edyvean et al. 1988), which can aggressively corrode steel in reinforced concrete. However, unless the seaweed is significantly aggregated in a closed system, the significance of this is likely to be low as high concentrations will be rapidly dissipated in open systems by waves and currents (Buenfeld 1984). Additionally, this will only be of detriment if the concrete facing is already deteriorated, and the rebar is exposed to seawater.

3.4.3 Marine invertebrates

3.4.3.1 Biophysical

Grazers, such as gastropod molluscs and particularly *Patella* spp. and *Littorina littorea*, are known to have a bio-erosive effect on soft rock, such as limestone (Schneider and Torunski 1983; Trudgill 1988;

Swantesson et al. 2006b). Through their feeding activity and excavation of 'home scars', limpets (*Patella vulgata*) on the shores of East Sussex, southeast England, were found to be responsible for lowering the chalk platform on average 0.15 mm per year depending on their density (Andrews and Williams 2000). Notably, high concentrations of calcium were present in faecal pellets of limpets also grazing on siliceous rock, suggesting much of the calcium was derived from the algae consumed, and not necessarily due to the ingestion of particulate rock. Various assumptions made about the faecal pellet method mean these results should be interpreted cautiously and possibly overestimate the bioerosive impact of limpet grazing. However, where soft rock and concrete surfaces have been agitated via grazing or home scar formation, they are likely to be more vulnerable to other weathering agents.

Bivalve molluscs can also contribute to concrete deterioration through the invasive nature of attachment structures such as byssal threads. Perez et al. (2003) used SEM and energy-dispersive X-ray spectroscopy to analyse concrete colonised by the freshwater golden mussel (*Limnoperna fortunei*) in Argentina where it is a non-native species. Results demonstrated the byssal threads, which the mussels use to remain attached to the substrate, penetrate the material surface and can cause fissures which increases the likelihood of water ingress and other erosive pathways. This was supported by similar findings by Yao et al. (2017), who found that colonisation of golden mussels on concrete reduced compressive strength. Concrete calcium content where mussels had colonised was reduced and both Perez et al. (2003) and Yao et al. (2017) deduced that the mussels leach calcium from the concrete for their shell growth. However, mussels derive calcium from the water and do not absorb it from the substrate (Ramesh et al. 2017). It is more likely that increased water ingress via byssal thread penetration in the concrete had led to the dissolution of calcium leachate.

Some boring bivalves, such as piddocks, secrete a substance that enables chelation, a chemical process that bonds molecules to metal ions and dissolves calcareous substrata in which the piddocks burrow. Other piddock and clam species burrow mechanically by abrading the rock with their shells (Trudgill and Crabtree 1987; Bromley and Heinberg 2006). In conjunction with abiotic weathering mechanisms, bivalve boring can represent a significant bioerosive risk to carbonate-based coastal infrastructure (Pinn et al. 2005; Moura et al. 2012; Coombes et al. 2014). Boring activity also occurs in concrete. Scott et al. (1988) recorded boring activity from polychaete worms, sponges and bivalve molluscs in tropical coastal limestone and concrete. This activity was concentrated where the limestone aggregate in the concrete matrix was densest, with the sponges avoiding the cement paste altogether.

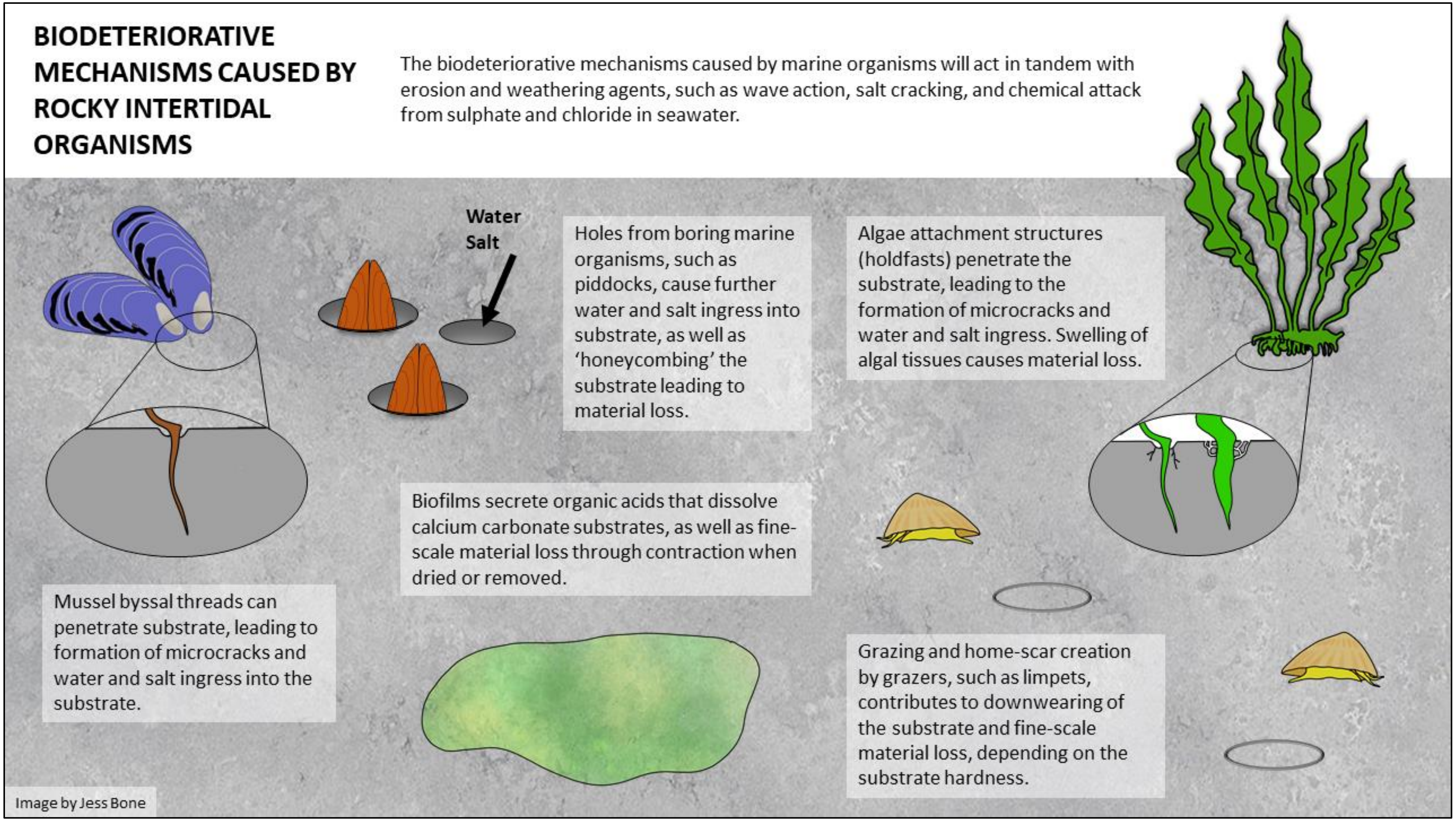
3.4.3.2 Biochemical

Carbon dioxide increases within seawater overnight as a result of respiration and the cessation of photosynthetic activity (Emery 1946; Trudgill 1976; Lundberg 1977; Moses 2002), which may increase dissolution of calcareous rock and cement nocturnally (Griffin et al. 1991; Sand 1997; Garcia-Pichel 2006). However, this is only likely to impact closed systems on a very minor local scale and is likely to have little impact on vertical concrete structures.

Table 2. The mechanisms by which rocky intertidal organisms may provide bioprotective or biodeteriorative effects.

	Biodeteriorative Action	Substrate Affected	References	Bioprotective Action	Substrate Affected	References
Microorganisms, biofilms, and extra-cellular polymeric substances	Solubilisation of metal ions and substrate decomposition/ secretion of organic acids	Limestone coastal rock	Eckhardt 1985; Flemming 1993; Schneider and Le Campion-Alsumard 1999; Cwalina 2008; Scheerer et al. 2009	Lichen hyphal penetration of pore spaces can inhibit the ingress of water and solubilizing chemicals	Tuff (igneous rock)	Garcia-Valles et al. 2003
	Creation of microscopic boreholes that can coalesce and lead to material loss	Granite, limestone, concrete	Schneider and Le Campion-Alsumard 1999; Coombes et al. 2011	Precipitation of insoluble material or promotion of cementation inhibiting further water ingress	Concrete, limestone, sandstone	Costa and Appleton 1999; Di-Bonaventura et al. 1999; Neville 2004; Gadd and Dyer 2017
	Expansion and contraction of cells through wetting and drying cycles leading to microcrack formation	Granite, limestone, concrete	Moses and Smith 1993; Gomez-Pujol et al. 2007; Coombes et al. 2011	Lichen coverage can retain moisture and reduce thermal fluctuations, protecting against weathering	Limestone, sandstone	Arino et al. 1995; Fiol et al. 1996; Carter and Viles 2003
	Enhancement of biological crust formation leading to precipitation of gypsum and brucite increasing risk of microcracks	Concrete	Costa and Appleton 1999; Neville 2004	Biofilms can form protective layer to retard salt penetration	Marine mortar	Lv et al. 2015a; Gao and Tang 2018
	Desiccation of biofilms leads to removal of mineral grains from substrate surface as biofilm contracts	Limestone, brick, mortar	Guillitte and Dreesen 1995			
Macroalgae	Penetration of attachment structures leads to fine scale material loss	Granite, limestone, concrete	Morrison et al. 2009; Hughes et al 2013a; Hughes et al. 2013b	Colonisation of macroalgae reduces the space vulnerable to biodeteriorative action from cyanobacteria	Limestone	Naylor and Viles 2002
	Expansion and contraction of surface penetrating tissues through wetting and drying cycles leading to particle disaggregation	Granite, limestone	Morrison et al. 2009	Algal turf reduces downwearing rates	Carbonate rock	Moura et al. 2012

	Secretion of organic acids can lead to chemical etching and may lead to dissolution of calcium	Calcareous rock, concrete	Welton et al. 2003; Jayakumar and Saravanane 2009; 2010; Jayakumar et al. 2011	Algal canopies moderated temperature extremes and buffered humidity variability	Concrete, limestone, mudstone	Coombes et al. 2013; Gowell et al. 2015; Scrosati and Ellrich 2018
				Crustose algae may provide a protective layer and cement loose material to the substrate surface	Sedimentary rock	Trenhaile 2017; Kennedy et al. 2019
Marine Invertebrates	Scouring of substrate via limpet feeding activity and home scar formation	Chalk coastal platforms	Andrews and Williams 2000	Barnacle cover inhibits other weathering mechanisms by forming a protective layer	Limestone, sandstone	Moura et al. 2006; Pappalardo et al. 2018
	Penetration of attachment structures such as byssal threads of mussels can lead to formation of fissures and reduction in compressive strength	Concrete	Perez et al. 2003; Yao et al. 2017	Barnacle, vermetid worm, mussel and oyster cover moderates thermal extremes and inhibits salt penetration by forming a protective layer. Barnacle and oyster secretions and adhesives reduce concrete porosity.	Limestone, granite, concrete	Risinger 2012; La Marca et al. 2015; Lv et al. 2015b ; McAfee et al. 2016; Coombes et al. 2017; La Marca 2017; McAfee et al. 2017; Chlayon et al. 2018; Jurgens and Gaylord 2018; McAfee et al. 2018; Lv et al. 2021; Lv et al. 2022
	Boring activity from piddocks, worms and sponges lead to material loss and further water ingress	Limestone, concrete	Scott et al. 1988; Pinn et al. 2005; Bromley and Heinberg 2006; Moura et al. 2012	Mussel and oyster beds dissipate wave energy	Unknown	Donker et al. 2013; Wiberg et al. 2019; Gonzalez et al. 2021; Baxter et al. 2022
				Piddocks secrete insoluble calcite within boreholes which inhibits further water ingress	Carbonate rock	Moura et al. 2012
				Honeycomb worms stabilise sediment, reducing abrasion and attenuating wave energy	Coastal rock	Naylor and Viles 2000; Braithwaite et al. 2006; White 2011
Entire sessile assemblages	Unknown	Unknown	Unknown	The build-up of organisms creates a physical barrier against other weathering mechanisms and salt penetration	Marine concrete	El-Hawary et al. 2000; Maruya et al. 2003; Kawabata et al. 2012; Georges et al. 2021



BIODETERIORATIVE MECHANISMS CAUSED BY ROCKY INTERTIDAL ORGANISMS

The biodeteriorative mechanisms caused by marine organisms will act in tandem with erosion and weathering agents, such as wave action, salt cracking, and chemical attack from sulphate and chloride in seawater.

Holes from boring marine organisms, such as piddocks, cause further water and salt ingress into substrate, as well as ‘honeycombing’ the substrate leading to material loss.

Algae attachment structures (holdfasts) penetrate the substrate, leading to the formation of microcracks and water and salt ingress. Swelling of algal tissues causes material loss.

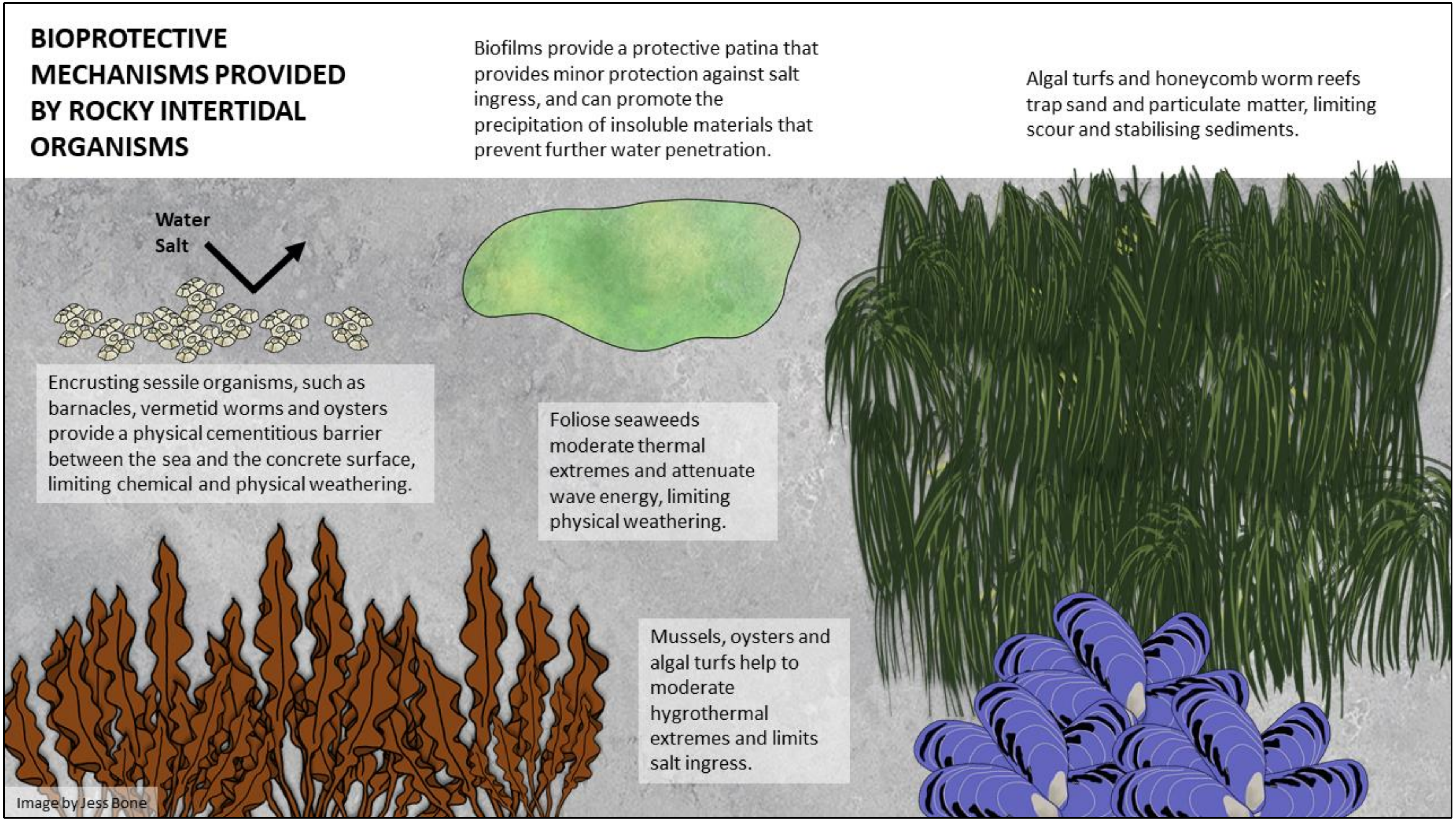
Biofilms secrete organic acids that dissolve calcium carbonate substrates, as well as fine-scale material loss through contraction when dried or removed.

Mussel byssal threads can penetrate substrate, leading to formation of microcracks and water and salt ingress into the substrate.

Grazing and home-scar creation by grazers, such as limpets, contributes to downwearing of the substrate and fine-scale material loss, depending on the substrate hardness.

Image by Jess Bone

Figure 3. The biodeteriorative action of rocky intertidal organisms. Note that while substrate is indicative of concrete, the magnitude of biodeteriorative action is dependent on the substrate material and spatiotemporal variations in population size of the organism. Not to scale



BIOPROTECTIVE MECHANISMS PROVIDED BY ROCKY INTERTIDAL ORGANISMS

Biofilms provide a protective patina that provides minor protection against salt ingress, and can promote the precipitation of insoluble materials that prevent further water penetration.

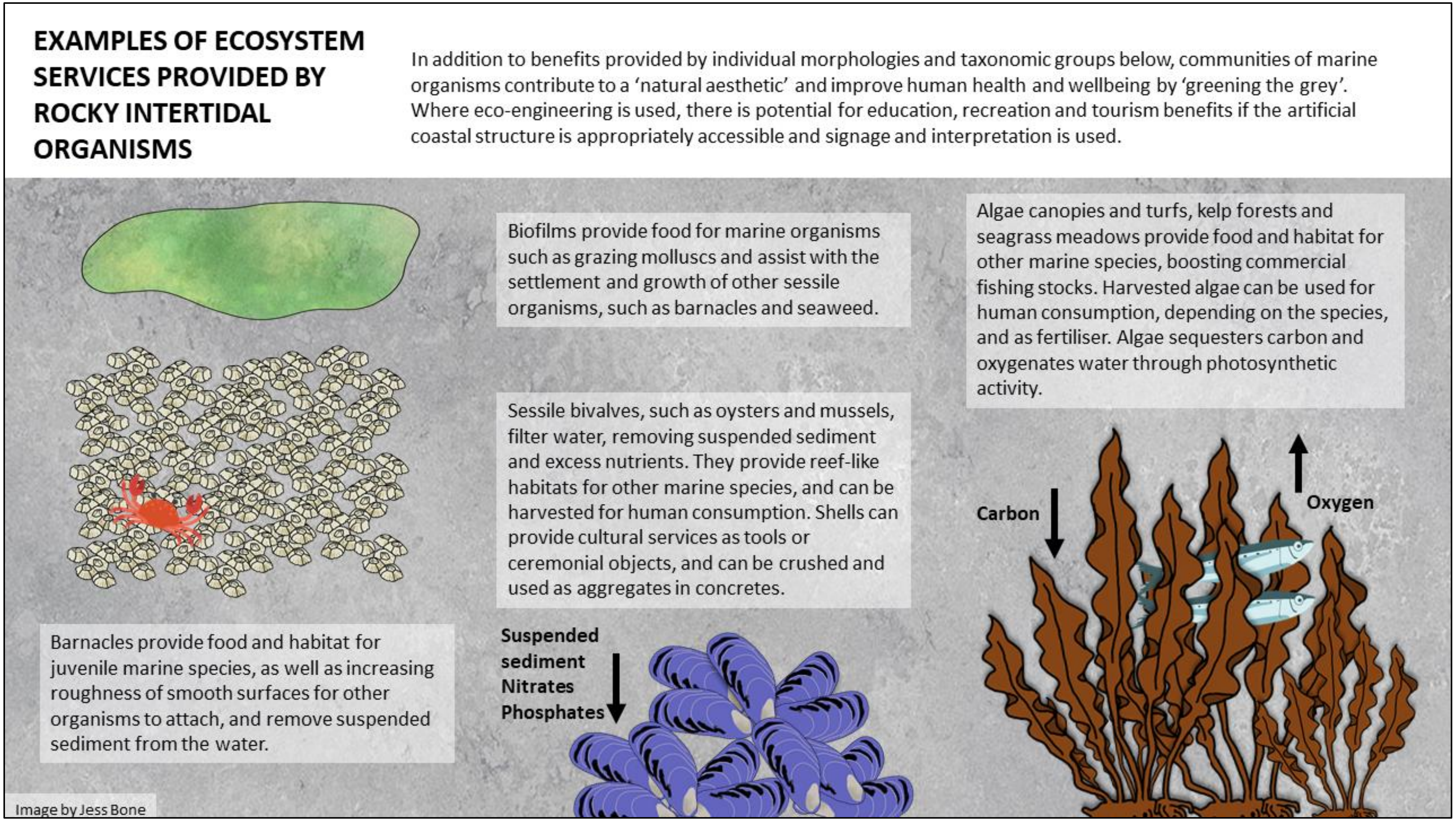
Algal turfs and honeycomb worm reefs trap sand and particulate matter, limiting scour and stabilising sediments.

Encrusting sessile organisms, such as barnacles, vermetid worms and oysters provide a physical cementitious barrier between the sea and the concrete surface, limiting chemical and physical weathering.

Foliose seaweeds moderate thermal extremes and attenuate wave energy, limiting physical weathering.

Mussels, oysters and algal turfs help to moderate hygrothermal extremes and limits salt ingress.

Figure 4. The bioprotective effects of rocky intertidal organisms. Note that while substrate is indicative of concrete, the magnitude of bioprotective benefits is dependent on substrate material, whether the organism is live, and the spatiotemporal variations in population size and density. Not to scale.



EXAMPLES OF ECOSYSTEM SERVICES PROVIDED BY ROCKY INTERTIDAL ORGANISMS

In addition to benefits provided by individual morphologies and taxonomic groups below, communities of marine organisms contribute to a ‘natural aesthetic’ and improve human health and wellbeing by ‘greening the grey’. Where eco-engineering is used, there is potential for education, recreation and tourism benefits if the artificial coastal structure is appropriately accessible and signage and interpretation is used.

Biofilms provide food for marine organisms such as grazing molluscs and assist with the settlement and growth of other sessile organisms, such as barnacles and seaweed.

Algae canopies and turfs, kelp forests and seagrass meadows provide food and habitat for other marine species, boosting commercial fishing stocks. Harvested algae can be used for human consumption, depending on the species, and as fertiliser. Algae sequesters carbon and oxygenates water through photosynthetic activity.

Sessile bivalves, such as oysters and mussels, filter water, removing suspended sediment and excess nutrients. They provide reef-like habitats for other marine species, and can be harvested for human consumption. Shells can provide cultural services as tools or ceremonial objects, and can be crushed and used as aggregates in concretes.

Barnacles provide food and habitat for juvenile marine species, as well as increasing roughness of smooth surfaces for other organisms to attach, and remove suspended sediment from the water.

Image by Jess Bone

Figure 5. Examples of some of the ecosystem services provided by the rocky intertidal organisms mentioned in this review. Not to scale.

3.5 Bioprotective effects

Bioprotection can be achieved via three main mechanisms: stabilisation of the substrate, microclimate mediation, and attenuating weathering and other deteriorative effects (Figure 4). Substrate stabilisation can involve the aggregation of particulate matter within biofilms, the secretion of insoluble precipitates on the substrate surface, such as oxalates, and the retention of sediment in algae. Buffering hygrothermal regimes and reducing the frequency of extreme temperature events can be achieved through the colonisation of sessile and epilithic biota, which mediates microclimate compared to bare substrate. Additionally, colonisation can mitigate against other weathering mechanisms, including chemical, physical and biological erosion.

Several studies that examine the bioprotective effect of entire sessile assemblages on coastal concrete can attest to its enhanced resistance to chloride ion (salt) penetration (Maruya et al. 2003; Kawabata et al. 2012; Georges et al. 2021). El-Hawary et al. (2000) compared epoxy-repaired concrete samples in the intertidal zone and in the lab and found that, unlike the lab samples, the field samples did not show degradation or a reduction in tensile strength during the entire 18-month study period. It was assumed that the thick build-up of sessile organisms protected the field concrete from exposure to seawater and the associated chemical deterioration. The majority of bioprotection studies tend to focus on specific morphology (e.g., biofilms, macroalgae) or individual species. While this adds validity to the bioprotection argument for that species or morphology, it does mean the dynamism of its existence through time and space and among wider assemblages is often not considered. However, it is important to demonstrate that although many species and life forms may cause biodeterioration, they often dichotomously demonstrate the ability to offer some level of bioprotection (Table 2).

3.5.1 Microorganisms, biofilms and lichen

There is extensive work quantifying the bioprotective effects of lichen on terrestrial cultural heritage (Carter and Viles 2003), some of which is reviewed here to provide wider context for marine application. Such bioprotective effects may be relevant in the supralittoral fringe (splash/ spray zone) on artificial coastal structures where marine lichens can be found (Ryan 1988).

Lichens and fungal biofilms have been found to enhance biomineralization (or bioremediation) in stone cultural heritage (Gadd and Dyer 2017). Many microorganisms are able to perform this ecosystem service by precipitating carbonates which result in cementation, or the formation of insoluble minerals known as microbially induced calcite precipitation (MICP) (Di-Bonaventura et al. 1999; Dittrich and Sibler 2010; Al-Salloum et al. 2017). Although some marine microorganisms are known to precipitate calcites and MICP is known to occur on concrete, no study to date directly attributes 'healing' of cracks in intertidal concrete from the natural colonisation of microorganisms or EPS communities. Lv et al. (2015a) demonstrated that marine bacteria could form protective biofilms by retarding the permeation of chloride and magnesium ions into ordinary Portland cement mortar and inhibiting OH^- leachate. This is supported by Gao and Tang (2018) who found that chloride penetration in concrete in the intertidal was reduced where biofilms were present. Additionally, the superficial invasion of hyphae into the pore spaces of limestone was found to reduce the ingress of water and solubilizing chemicals (Garcia-Valles et al. 2003). On lab based mesocosm experiments conducted at >90% humidity, Fiol et al. (1996) found that dissolved and particulate loss of limestone material was greater on bare rock than lichen covered rock. This was supported by Carter and Viles (2003) who found that lichens retain moisture and subsequently reduce thermal stress and the magnitude of thermal fluctuations on substrate surfaces. Arino et al. (1995) addressed the balance of biodeterioration versus bioprotection when comparing areas of lichen covered Roman pavement with bare pavement in Spain. Although lichen colonisation did show biological weathering to the pavement surface, these biodeteriorative effects were deemed to be slower acting than abiotic weathering, which showed significant impacts in bare areas not covered by lichen. Although this evidence suggests that lichens may play a similar role in the supralittoral zone, no studies to date have clearly demonstrated this. However, terrestrial lichens show promising bioprotective properties, which should encourage further study of lichens and similarly structured life forms in the intertidal to confirm this.

3.5.2 Plants and macroalgae

Plants and macroalgae are already well known for their bioprotective effects on a larger spatial scale, particularly mangroves, seagrass meadows, saltmarshes and kelp beds, due to wave attenuation (Mazda et al. 1997; Massel et al. 1999; Quartel et al. 2006; Chen et al. 2007; Bradley and Houser 2009; McIvor et al. 2012; Anderson and Smith 2014; Horstman et al. 2014; Tambroni et al. 2016), storm mitigation (Moller et al. 2014; James et al. 2020), and sediment retention (Adame et al. 2010) and accretion (Gacia et al. 1999). There is mounting evidence that macrophytes also protect substrata on a smaller, localised spatial scale particularly via microclimate mitigation and acting as an 'umbrella', which has been observed in both terrestrial plants (Sternberg et al. 2010) and macroalgae.

Naylor and Viles (2002) found that biodeteriorative effects of cyanobacteria colonisation and weathering were reduced once macroalgae had established on blocks of limestone installed on the rocky shore of Falsarna, Crete. Filamentous and foliose algae on exposed limestone blocks appeared to limit other biodeteriorative forces and macroalgal abundance was inversely related to bioerosion from cyanobacteria at the study close, suggesting that macroalgal colonisation provides some level of bioprotection.

Moura et al. (2012) visited the Algarve coast, Portugal, looking at downwearing rates of two carbonate rock platforms. Downwearing rates were lower on substrate covered in algal turf compared to bare rock, suggesting that macroalgae offered bioprotective benefits via wave attenuation and trapping sand that would otherwise scour the rock surface.

The brown canopy-forming algae *Fucus* spp. was found to moderate the range and maxima of daily summer temperatures on concrete and limestone artificial coastal structures in the southwest of England compared to control areas that had been cleared (Coombes et al. 2013). Short term temperature and humidity variability was reduced by up to >70% under *Fucus* canopies during low tide. The amelioration of wetting and drying cycles associated with tidal regimes could represent a reduction in salt weathering, and the reduction in direct solar radiation and thermal stress could limit other weathering effects. Gowell et al. (2015) compared the hardness and surface condition of mudstone with artificial macroalgal canopy compared to an uncovered control under simulated intertidal conditions. It was found that the artificial macroalgae buffered microclimatic fluctuations by modifying temperature and humidity at the mudstone surface. Compared to the uncovered control, the covered mudstone lost 80% less debris and did not undergo as great a reduction in hardness. The artificial algae canopy also retained moisture and shaded the mudstone surface, reducing the frequency of salt crystallization events. Reducing the salt ingress in the material surface would be a key benefit in concrete artificial coastal structures as chloride attack on steel reinforcement in concrete is a leading cause of structural decay (Neville 2011). *Ascophyllum nodosum* canopies were found to insulate intertidal substrate in winter on the Atlantic Canadian coast with temperatures on bare substrate up to 10 degrees lower than canopy-covered substrate (Scrosati and Ellrich 2018). It can be inferred that algal cover has the potential to therefore provide insulation to substrata throughout the year in temperate regions, buffering thermal regimes which may subsequently ameliorate other deteriorative forces. Crustose algae may also play a key bioprotective role. Kennedy et al. (2019) observed that the red coralline algae *Lithophyllum incrustans* appeared to protect an intertidal platform on the Yorkshire coast, UK, from weathering. Coralline and encrusting algae on artificial coastal structures may perform similar roles by providing a protective patina (Trenhaile 2017).

3.5.3 Marine invertebrates

The marine invertebrates that have thus far been determined to play a bioprotective role are generally gregarious and sessile, and include barnacles, calcareous tube-building worms, mussels and oysters. The calcareous structures formed by these species (shells, tests) form a solid and rough layer on the substrate surface.

The rocky shore topography of Algarve, Portugal was characterised by Moura et al. (2006) who observed that barnacle dominated rock platforms were typically very irregular as areas not covered by barnacles were physically weathered by wave action. It was concluded that barnacle cover provided some protection from wave erosion and thus contributed to the topographical heterogeneity of the shore platforms. Varying percentage cover of *Chthalamus* sp. barnacles were compared on limestone, granite and marine-grade concrete substrates under simulated intertidal conditions by Coombes et al. (2017). Subsurface peak temperatures were reduced by >5 °C in concrete with near total coverage of barnacles compared to bare concrete with no barnacle cover. There was a statistically significant negative relationship between thermal breakdown and barnacle cover. Chloride ion migration was lower in materials covered by barnacles, suggesting a reduction in salt ingress. Additionally, evaporative cooling occurred with barnacle covered materials, due to the loss of water retained in the empty barnacle tests. The thermal regime observed here may differ *in situ* with live barnacles as the moisture is retained within the tests at low tide and so the evaporative cooling may be reduced. However, the inhibited chloride ion migration under barnacle cover may hint at bioprotective effects which are supported by La Marca et al. (2015), particularly on reinforced concrete where steel rebar is vulnerable to corrosion. Pappalardo et al. (2018) conducted a manipulative field experiment on the northwest coast of Italy by comparing the hardness and weathering of areas colonised by *Chthalamus* sp. and bare areas scraped clear. After four months, the bare exposed rock was less hard and showed more weathering than barnacle covered rock. Similar effects have been observed in calcareous tube-building worms. Vermetid worm encrustations were found to reduce peak temperatures of rock compared to uncolonised bare rock in a mesocosm experiment by La Marca (2017), in addition to reducing salt ingress. Chlayon et al. (2018) found that the barnacle *Chthamalus challeneri* improved concrete durability on an intertidal concrete jetty by sealing microcracks and limiting chloride diffusion. Lv et al. (2022) found barnacles provided concrete in the marine environment ‘three lines of defense’ with their tests, adhesive and the penetration of their adhesive, enhancing resistance of water absorption and chloride ion penetration and improving the concrete durability.

Mussel (*Mytilus californianus*) aggregations on the Washington, north Californian and south Californian coast, US, were found to functionally eliminate lethal and sublethal temperatures for intertidal biota compared to bare rock, regardless of tidal elevation and latitude (Jurgens and Gaylord 2018). Gonzalez et al. (2021) and Baxter et al. (2022) demonstrated that mussels perform a similar bioprotective function on coastal substrates as barnacles. Gonzalez et al. (2021) found that experimental removal of mussels (*Brachidontes rodriguezii*) on a shore platform on the Argentinian coast led to a 10% decrease in surface hardness after 5 months. Baxter et al. (2022) supports these findings by comparing mussel-covered (*Mytilus edulis)* intertidal rock on the English coast with bare rock. It was found that mussel-covered rock was significantly harder than bare rock, with mussels moderating microclimate regimes, water motion and turbulence at the rock surface. Oysters are also known to facilitate thermal buffering at the substrate surface by shading and trapping moisture in the interstices between shells (McAfee et al. 2016; 2017; 2018) and thus it can be inferred they may play a similar bioprotective role (Risinger 2012). Oyster cementation inhibits chloride ion permeability, enhances concrete durability, reduces pore structure at the concrete surface (Lv et al. 2015b), reduces water absorption and enhances resistance to carbonation (Lv et al. 2021). Additionally, both oyster (Wiberg et al. 2019) and mussel (Donker et al. 2013) beds are known to dissipate wave energy in the intertidal zone. Unlike mussels and barnacles, the cementitious secretions of oysters to facilitate attachment are predominantly inorganic and resistant to acid solubilisation (Burkett et al. 2010; Tibabuzo Perdomo et al. 2018) and therefore may persist as a protective biogenic layer on the substrate following death.

The honeycomb worm (*Sabellaria* spp.) is a reef-building gregarious organism that settles on hard intertidal substrates and can produce reefs several centimetres in height (Naylor and Viles 2000) and several metres in diameter. Their bioconstructions are comprised of tubes made from sand grains cemented together via secretions, which play a potentially bioprotective role by stabilising sand that would otherwise abrade hard substrates and attenuating wave energy (White 2011). Braithwaite et al. (2006) noted that expansive honeycomb worm reefs had buried subsea pipelines.

3.6 Discussion

It is evident that virtually all marine biota possess a biodeteriorative capacity for intertidal substrata, often via multiple mechanisms. However, many of these species, such as macroalgae and barnacles, also jointly offer bioprotective effects. It has been noted by several authors (Naylor et al. 2002; Naylor et al. 2005; Carter and Viles 2005; McIlroy de la Rosa et al. 2012; Favero-Longo and Viles 2020) that biodeterioration and bioprotection should not be viewed as conflicting, dichotomous, isolated positions; rather they are two ends of the same scale with both acting in tandem. The biodeteriorative effects inflicted by a given species may be outweighed by bioprotective effects that inhibit and retard more severe and faster acting weathering as observed with lichen (McIlroy de la Rosa et al. 2014) on Angkor temples by Bartoli et al. (2014). The hyphal penetration of lichen was deemed overall less deleterious than if the lichen was absent and the stone substrate was exposed to direct sunlight and weathering.

Determining if a given species or assemblage is providing overall net bioprotective or deteriorative effects is dependent on many complex, dynamic and interrelated biotic and abiotic factors that are temporally and spatially variable. For example, the net bioprotective benefits of barnacles depend on their percentage cover, whether they are live (Coombes et al. 2017), their age, and their attachment method as some barnacle species etch into the substrate and seal their tests directly to the substrate (Donn and Boardman 1988; Bromley and Heinberg 2006) and other species use adhesive (Pappalardo et al. 2018; Liang et al. 2019). Naturally, their percentage cover will also vary over time and space. Further, biodeteriorative and bioprotective forces are mediated by the material they are acting on and at different scales, with different lithologies and concretes more or less vulnerable to different biogenic processes than others (Coombes 2014; MacArthur et al. 2020). As noted by Coombes et al. (2013), wave-driven erosion is likely to have a greater impact on less durable materials, such as carbonate rock, than limpet grazing.

Facilitative bioerosion alters the properties of the substrate material, for example by reducing material strength or exploiting discontinuities and joint planes, making it more vulnerable to weathering and erosive forces (*sensu* Naylor et al. 2012). Therefore, consideration must also be given to the management of fouling organisms as the removal of colonisation may accelerate other deteriorative forces. The removal of lichen on Angkor temples was considered inadvisable, as the damage caused by the lichen hyphae increased the substrate's vulnerability to weathering once the lichen was removed, a consideration supported by McIlroy de la Rosa et al. (2012). This may also be observed on concrete coastal structures where the removal of barnacles and macroalgae, which would require abrasive action such as power washing (Hughes et al. 2013b), may expose and exacerbate surface damage caused by biodeterioration (Pappalardo et al. 2018), such as expanded pores and microcracks. Such damage may permit deeper water and salt ingress and expose the substrate to the deleterious hygrothermal regimes recorded on bare rock (McAfee et al. 2016; Coombes et al. 2017; La Marca 2017; McAfee et al. 2017; Jurgens and Gaylord 2018; McAfee et al. 2018; Pappalardo et al. 2018). Additionally, it should be noted that management that entails the periodic removal of biofouling assemblages may lead to the domination and spread of non-native invasive species and loss of biogenic habitat and biodiversity. Coombes et al. (2013) suggested that loss of macroalgal cover shifted the substrate from a stable state to an unstable state, due to not only the return of aggressive mechanical weathering but enhanced ecological stress. McIlroy de la Rosa et al. (2012) demonstrated with conceptual modelling that instability and rapid topographical change occurs on substrates following the death and decay of epilithic lichen. Subsequently, bioprotective species and assemblages can provide crucial stabilisation for substrata. The bioprotective effects of some species are however, easily outweighed by their net biodeteriorative effects. For example, boreholes in the rocky coast of Portugal were stabilised by insoluble calcite precipitated by the piddocks which protected against further water ingress but contributed to significant material loss (Moura et al. 2012). As opposed to weighing up the biodeteriorative versus bioprotective effects of that species or assemblage, the effects of their absence (and particularly removal) should also be included in the analysis. As stressed by Coombes (2014), biogenic and physico-chemical processes occur synergistically and it is the cumulative impact, or total weathering outcome (Hall et al. 2012), of these interactions that should be assessed. Liu et al. (2022, in press) proposed a bioprotection ratio for biofilms heritage stone monuments that considers natural weathering as all structures are impacted by this.

When attempting to determine if a species or assemblage offers a net bioprotective gain or biodeteriorative loss, acknowledgement of the additional ecosystem services offered by it should be considered (Figure 5). For example, carbon dioxide sequestration and water filtration (Layman et al. 2014), habitat and food provision (Vaughn 2018), contribution to 'natural' aesthetic (Fairchild et al. 2022), and cultural and socio-economic benefits (education, tourism, recreation, fisheries industry) may be provided (Naylor et al. 2017) and could tip the scales in favour of encouraging or retaining existing colonisation (Coombes and Viles 2021). Ultimately, a bioprotective assemblage is not guaranteed as colonisation of a substrate is dependent on spatiotemporally variable factors, such as latitude, climate, orientation, aspect, substrate material, larval supply, predator-prey dynamics, competition, and habitat and food availability. As noted by Gadd and Dyer (2017) there is no guarantee that target species will colonise or perform in the expected and desired manner. However, promoting a diverse community on artificial coastal structures is likely to increase the number of bioprotective species and their percentage cover, tipping the scale towards net bioprotection and away from biodeteriorative species such as cyanobacteria (Naylor and Viles 2012). Typically, artificial coastal structures do not represent diverse habitats and their communities are not analogous to natural communities (Connell and Glasby 1999; Chapman 2003; Moschella et al. 2005; Vaselli et al. 2008; Pister 2009). However, through considered eco-engineering and bioreceptivity enhancement (Bone et al. 2022a) and with appropriate consultation with marine ecologists, artificial coastal structures can host a diverse suite of species with high percentage cover that offer more than just substrate bioprotection (Chapman and Underwood 2011; Dafforn et al. 2015; Bishop et al. 2017; O'Shaughnessy et al. 2019). Coombes et al. (2013) opined that features enhancing the recolonisation of canopy-forming macroalgae, such as artificial rockpools (Hall et al. 2019) and mud pools (Bone et al. 2022b), on highly disturbed artificial coastal structures should be an eco-engineering priority. Conversely, Chlayon et al. (2020) argued that sessile crusts, consisting of calcareous invertebrates, should be prioritised over algae after comparing the bioprotective and biodeteriorative properties of both on concrete surfaces.

When considering how to facilitate bioprotective species, it is worth first understanding how their colonisation may work collaboratively with the artificial coastal structures. For example, vertical seawalls that provide flood defence could benefit from colonisation of canopy-forming macroalgae, such as fucoids, to attenuate wave energy and associated risks (Coombes et al. 2013), such as overtopping. By permitting the colonisation or continuation of existing assemblages, maintenance costs associated with biofouling management will reduce, although condition checks may subsequently increase. Naturally, where colonisation poses risk to the public (e.g., algae on pathways) or loss of function (e.g., power plant water intake), the removal of biofouling communities must continue.

In a warming, acidifying ocean, bioprotection may play an important role in creating a physical barrier between alkaline substrata and acidic seawater. It has been demonstrated that concrete may provide superior substrate for algal turf cover and photosynthetic efficiency compared to granite under acidification scenarios (Davis et al. 2017). The dissolution of $CaCO_3$ was deemed a key factor in concrete performance as substrate in acidifying oceans. This is supported by Mos et al. (2019) who found that although settlement rates of juvenile tropical sea urchins (*Tripneustes gratilla*) were lower on concrete compared to granite and greywacke (a hard sandstone) under simulated ocean acidification scenarios, post-settlement juveniles on concrete were larger and had higher survival rates after two weeks. The alkali leachate was thought to buffer the low pH conditions of the ambient seawater, creating favourable conditions. Given the potential bioprotection offered by calcareous organisms, such as barnacles, oysters, and mussels, that would benefit from alkali buffering in acidifying oceans, there is justification for enhancing bioreceptivity of concrete artificial coastal structures. Further, increasing air temperatures and frequency of storms due to climate change may enhance the net bioprotective capabilities of intertidal organisms, such as buffering hygrothermal regimes and attenuating waves.

Further research should determine the bioprotective effects of whole assemblages *in situ*, as studies thus far have focussed on single taxa or functional group but should assess percentage cover of key morphologies/ taxonomic groups to help apportion variations in biogenic impacts to different organisms. Additionally, further work should also consider how bioprotection benefits may present subtidally and in tropical and polar regions, as the deleterious effects associated with intertidal wetting/ drying cycles and salt crystallization will differ in these contexts. Mesocosm experiments should also shift focus to how colonisation may inhibit deleterious effects associated with ocean acidification and warming scenarios by creating a physical barrier at the water substrate interface and preventing dissolution of concrete material. Such studies should apply caution and avoid bias towards reporting bioprotective results as it is evident from this review that colonisation is dually biodeteriorative and bioprotective. The interests of coastal managers and asset owner should be considered and maintained, as their collaboration is fundamental to permit new or continued colonisation on artificial coastal structures. Consequentially, it would be beneficial to translate quantitative results of bioprotection studies (e.g., substrate hardness, material loss) into metrics that are useful for coastal asset owners and managers, such as time added to a structures serviceable life, or monetary value.

3.7 Concluding remarks

The 'biodeterioration vs. bioprotection' debate has been on-going in terrestrial and aquatic fields for several decades, but it is evident that the reality of a species or assemblages' net impact is more nuanced and spatio-temporally dynamic. Evidence shows that biological colonisation of artificial coastal structures can cause deleterious effects that affect superficial topography and enhance abiotic weathering. Conversely, common intertidal species have been shown to buffer microclimatic conditions and retard weathering on coastal substrates and their colonisation may provide net bioprotective benefits compared to bare hard substrate. In an era where greater attention is being paid to mitigating habitat loss and reducing the carbon footprint of artificial coastal structures, bioprotection is another ecosystem service provided by marine species that can support the argument for eco-engineering and enhancing bioreceptivity. Further research is required to reinforce the conclusions of existing research in different contexts; particularly future casting under climate change scenarios, and care should be taken to communicate this to coastal asset owners and managers in a meaningful and useful way.

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Chapter 4 – Inconsistent bioreceptivity of three mortar mixes in subtidal sites

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4.1 Abstract

Concrete is extensively used in coastal engineering and development which, in addition to its high carbon footprint, threatens intertidal habitats and ecosystems. Eco-engineering addresses this by designing habitat features into coastal infrastructure. The chemical bioreceptivity of cement has been shown to vary, but ordinary Portland cement is generally considered to be the least bioreceptive.

In this study, we compare two low carbon mortars (a natural, single source cement (VP), and an ordinary Portland cement/ ground granulated blast furnace slag blend (GGBS)) with an ordinary Portland cement-based control mix (OPC). The three mortars were made into smooth blocks which were secured to crates and deployed subtidally in two estuary sites on the south UK coast for 1 year. At 3-, 6- and 12-months intervals a crate was recovered from each site and species abundance, biomass and assemblage composition were determined.

After 12 months, the VP mortar was significantly more species rich than both the OPC control and GGBS mortar, and organisms were significantly more abundant (numeric counts only), though this varied by mortar and site. However, OPC controls showed significantly higher percentage cover of biota than both low carbon mixes in both harbours. Overall, the GGBS mortar showed the least bioreceptivity of all three mortars. The primary chemical bioreceptivity of these mortars may perform inconsistently and other factors that impact bioreceptivity and primary succession, such as rugosity, should be considered when designing ecological enhancements. Sustainability of materials, such as opting for low carbon cements, should also be a priority.

4.2 Introduction

Concrete is a durable, cost-effective and versatile material which currently has no functional substitute for the construction of coastal infrastructure in the marine environment (Scrivener 2014; Alexander and Nganga 2016). Concrete is used in the majority of coastal infrastructure (Bijen 1996; Lukens and Selberg, 2004; Kosmatka et al. 2008), and in some areas 90% of coastline is developed (Cencini 1998). As coastal infrastructure proliferates (Dugan et al. 2011; Duarte et al. 2013; Duarte 2014), so too does the presence of concrete in intertidal and subtidal ecosystems, threatening coastal ecosystems (Bugnot et al. 2021). The carbon footprint of concrete construction is also significant with the cement industry contributing ≥5% to global CO_2 emissions (Worrell et al. 2001). Concrete is also considered an ecologically deficient analogue for natural hard substrate, such as rock (Connell and Glasby 1999; Chapman 2003; Moschella et al. 2005; Vaselli et al. 2008; Pister 2009; Bulleri and Chapman 2010). To address this ecological deficit in coastal infrastructure, ecological enhancement (also known as eco-engineering) is used to provide habitat for marine biota that was otherwise absent (Naylor et al. 2017; Strain et al. 2018; O'Shaughnessy et al. 2020; Evans et al. 2021). By virtue of the associated biological communities, ecological enhancement may also provide ecosystem services (Chapman and Underwood 2011; Strain et al. 2018), such as water filtration (Vozzo et al. 2021), improved fisheries (Chowdhury et al. 2021), and substrate bioprotection (Bone et al. 2022a).

Primary bioreceptivity is the propensity a material has for biological colonisation by virtue of its chemical composition and physical properties (Guillitte 1995). Extensive investigations have been made into the primary bioreceptivity of cementitious materials (Sanmartin et al. 2021; Bone et al. 2022b), particularly in more recent years as coastal engineers and practitioners incorporate ecological designs into their coastal structures and seek to encourage biocolonisation. This often involves modifying the binder (Perkol-Finkel and Sella 2014; Huang et al. 2016; McManus et al. 2018; Morin et al. 2018; Hayek et al. 2020; 2021; Ly et al. 2021 Natanzi et al. 2021) used to alter the surface chemistry, pH, albedo, hygroscopy and porosity to increase bioreceptivity and attract marine organisms more readily than unmodified industry standard binder. To address the growing demand for ecological enhancement, several companies now produce commercial off-the-shelf subtidal or intertidal units as standalone, integrated or retrofit options (Perkol-Finkel and Sella 2015; Hall et al. 2018; Perkol-Finkel et al. 2018; Evans et al. 2019; Bone et al. 2022c; Sella et al. 2022; Bishop et al. 2022; Drakard et al. 2023; Hickling et al. 2023). Examples of these can be found on coastlines worldwide, but focussed predominantly in North America, Europe and Australasia (Strain et al. 2017). Often these units comprise part of a larger coastal defence structure (Tschirky et al. 2018; Salauddin et al. 2021), and so they must be multipurpose, being robust and durable enough to ensure a long service life and be habitable for the marine life they hope to host. Several studies have been conducted on the concrete matrices used in these commercial products, comparing them with ordinary Portland cement (OPC)-based controls, to ensure they meet industry standards and provide a comparatively optimised surface for biocolonisation (Perkol-Finkel and Sella 2014; Hickling et al. 2022). Similar research has been conducted on concretes for use in artificial reef research pilots (Georges et al. 2021; Ly et al. 2021; Vivier et al. 2021). It is evident from these works that the materials used in ecological enhancements should be justified from an engineering, environmental (low carbon) and ecological perspective, in addition to the structure's architecture. Further, this evidence is crucial to ensure continued incorporation of habitat features in coastal and marine construction and to maximise the economic value of commercially available ecological enhancements. It was therefore prudent to determine how lower carbon cements may impact bioreceptivity and compare with OPC, given their increasing use in commercial ecological enhancements. Additionally, there is a paucity of field experiments quantifying *in situ* biocolonisation of concrete mixes, particularly in the subtidal environment and for longer than a few (≥3) months. We aimed to determine how bioreceptive lower-carbon cements (Vicat Prompt and GGBS) were compared to control cement (OPC) and how this changed over time with the following hypothesis:

The lower-carbon mortar blocks will be more bioreceptive than the OPC control mortar blocks after 12 months.

4.3 Method and materials

4.3.1 Mortars

Two low carbon mortars (Mortar 1 'GGBS' - CEMIII/B cement, 60% ground granulated blast furnace slag by Ecocem Ireland Ltd., 40% ordinary Portland cement; mortar 2 'VP' - a natural Roman binder, brand name Vicat Prompt) were trialled alongside a control mortar comprised of 100% ordinary Portland cement ('OPC', CEMI 52.5R cement, brand name Blue Circle Snowcrete).

Ordinary Portland cement is used in the construction of over half of artificial coastal structures globally (Lukens and Selberg, 2004; Perkol-Finkel and Sella 2014) but has a high pH of 12-14 (Taylor 1990; Manso et al. 2015). Consequently, initial biocolonisation of concretes with a high OPC content can be delayed, due to the prohibitively hostile surface conditions (Grant 1982; Nandakumar et al. 2003). Carbonation reduces the surface pH to 9-10 (Taylor 1990), permitting biological growth (John 1988; Manso et al. 2015), but the speed of this process is variable depending on environmental conditions (Hayek et al. 2020). OPC has also performed poorly compared to other binders (Manso et al. 2014; Perkol-Finkel and Sella 2014; Manso and Aguado 2016) but not exclusively (McManus et al. 2018; Veeger et al. 2021). The production of OPC requires high temperatures (≥1500°C), which requires significant energy input to achieve and is estimated to consume 2% of global primary energy consumption (Worrell et al. 2001). CO_2 is also produced during the clinker production, which involves the decomposition of limestone, and accounts for approximately half of the CO_2 emitted in the cement production industry (Worrell et al. 2001).

The carbon footprint of concrete may be reduced through the partial replacement of OPC by an alternative binder (Schneider et al. 2011) such as ground granulated blast furnace slag (GGBS), a recycled by-product of iron ore extraction (Neville 2011). In concrete containing a blend of OPC and GGBS (CEMIII), the pH is lowered (Guilbeau et al. 2003; Park and Tia 2004) and initial colonisation may improve (Hayek et al. 2020). GGBS may be used to replace OPC by up to 85% (British Standards Institute 2011). GGBS may enhance bioreceptivity in the marine environment, but results can be context dependent. After 1 month of intertidal exposure on County Meath coast, Ireland, Natanzi et al. (2021) found that OPC/ GGBS 50-50 mix concrete had greater microalgal biomass in a sheltered environment, but there was no difference between 100% OPC-based concrete and 50% OPC 50% GGBS concrete in the exposed environment. Following 7 weeks of subtidal deployment in Plymouth Sound, UK, McManus et al. (2018) found that the 100% OPC-based concrete tiles had the greatest native macro-fouling species richness compared to tiles containing 24% GGBS. For GGBS to have a significant impact on bioreceptivity, McManus et al. (2018) recommended that GGBS should replace OPC by a significant amount, which Natanzi et al. (2021) suggested should be at least 50%.

Vicat Prompt cement is a natural (single source) binder that has been used in the construction and repair of historic masonry and maritime structures for >100 years thanks to its durability, strength, seawater resistance and appropriate aesthetic (Gosselin et al. 2012; Baxter et al. 2022). It is more environmentally beneficial with a lower carbon footprint than artificial (blended) Portland binders, as it is fired at a lower temperature (between 800°C and 1200°C) and is very similar in composition to lime (Vicat 2003). It is used in the manufacture of commercially available ecological enhancements throughout Europe (personal communication Artecology 2020).

The mortar mixes comprised of binder at a 1:1 weight ratio with silica sharp sand (0-4 mm, sourced from Travis Perkins PLC.), and 1:2½ cement to water weight ratio. A retarder was added to the VP mortar according to manufacturer guidance to increase the setting time enough to pour the mixture into the moulds. The mortar mixes were hand-mixed and poured into silicone moulds to aid release of the cast blocks and avoid use of releasing agent. The silicone moulds produced mortar blocks with the dimensions 20 x 40 x 80 mm. Sixty replicate blocks were produced for each mix and cured in an indoor environment for the appropriate minimum time as recommended for each cement. Once cured, twelve replicates of each blend were fixed to a plastic crate, elevated and separated by 10 mm width plastic trunking (Figure 7) to permit adequate water flow around each block and avoid biotic contamination with its neighbour (*sensu* Ly et al. 2021). Each block had a “textured” side from the unfinished, exposed mortar following pouring, with all other sides smooth from their contact with the silicon moulds. To prevent surface texture from confounding the results, the “textured” side was placed face down on the crate using 3 mm cable ties and this face was not included in data collection. A total of six crates were set up with three crates deployed at each of the two study sites in June 2021; Poole Harbour, Dorset, UK (50.708745, -1.9863721, What3Words rungs.safely.range) and Hamble Harbour (50.852506, -1.3079996, What3Words bought.haggis.desiring), Hampshire, UK (Figure 6). The crates were deployed subtidally, suspended at least 1 m from the surface and seabed (*sensu* Ly et al. 2021). Both harbours are sheltered, muddy estuaries with crates suspended from either a jetty (Poole Harbour) or a pontoon (Hamble Harbour) with permission from harbour authorities and asset owners. A short video briefly showing block production and crate deployment may be viewed here (youtube.com/shorts/UnsPqyXc-Ko).

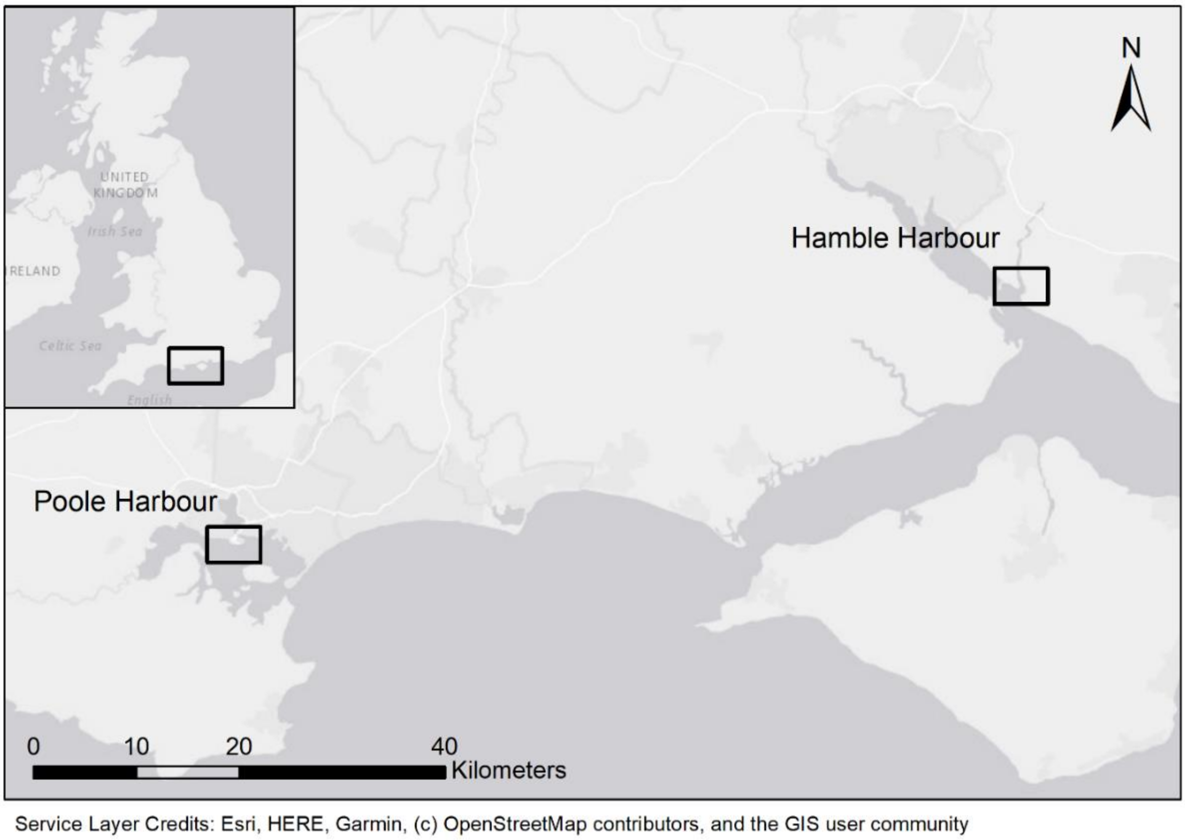


Figure 6. The deployment site locations of Poole Harbour and Hamble Harbour on the south coast of the UK.

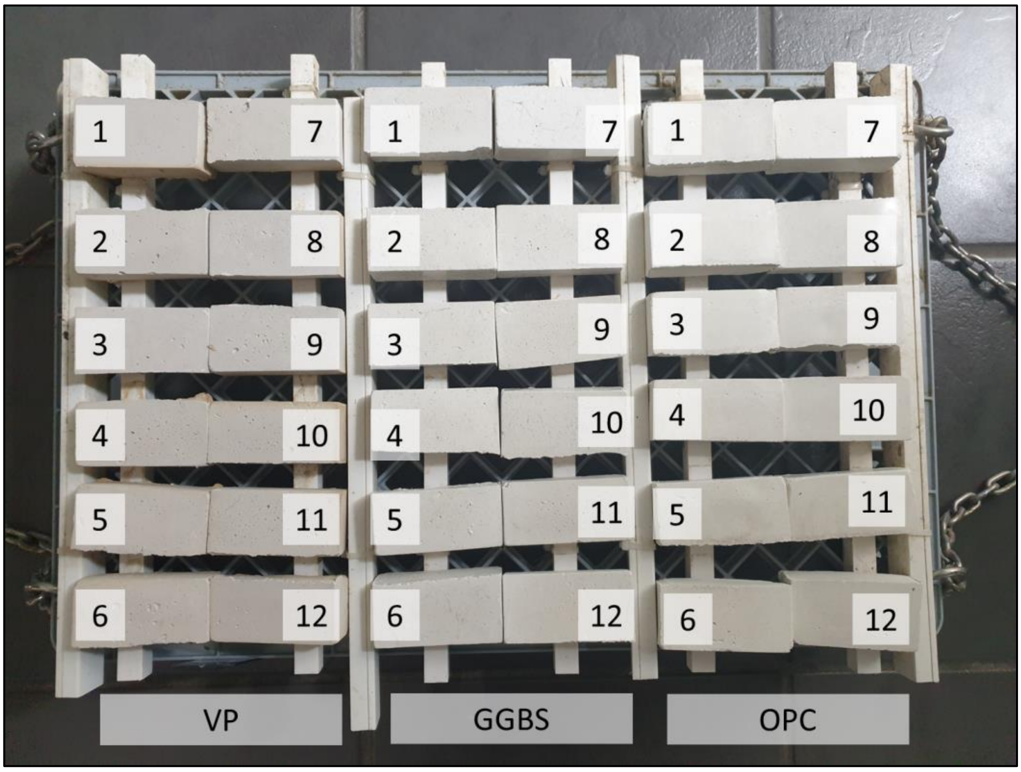


Figure 7. An example of how each of the six crates were set up. Numbers denote mortar block replicate.

4.3.2 Surveying

Colonising biota was assessed following the collection of one crate from each site at 3-, 6- and 12-month intervals and transported to the lab the same day (24 replicates of each mortar mix). High resolution photographs were taken of each block side (Sony A7R3 CDC camera, Sony FE 24-70mm zoom lens), except the ends and "textured" surface face. The percentage cover and numeric counts of organisms was visually estimated (Table 3 for breakdown) on these same block sides to species level wherever possible, while biota from the block ends and textured face was discounted. Sessile organisms were gently scraped off the block surfaces using a spatula (*sensu* Pappalardo et al. 2018). To determine biomass, organisms removed from each block were weighed when wet, dried in an oven at 100°C for 24 hours, or until a constant weight was achieved, and then dried in a chamber furnace at 500°C for 12 hours to obtain ash free dry weight (g) (Luczack et al. 1997; Heiri et al. 2001).

4.3.3 Data analysis

To test for statistically significant differences in abundance, species richness, and biomass between the mortar formulas over time within each site, generalised linear models (GLM) were run in R Studio (Version 1.2.1335) with 'mortar' (OPC, GGBS and VP) and 'interval' (3-, 6-, and 12-months) as fixed factors. Pairwise tests were run using the "emmeans" package (Lenth 2021). Poisson distribution was used on count data with equal variances and Quasi-Poisson distribution was used when the data were over dispersed (Crawley 2012).

Plymouth Routines in Multivariate Ecological Research (Primer-e v.7) was used to perform individual PERMANOVAs to test for differences in sessile assemblage structure between mortar, interval and interactions between mortar and interval using species abundance data (Anderson 2005). The data were square root transformed prior to use, to avoid the weighting of common species over rare. A Bray–Curtis resemblance matrix was used with 9999 permutations and PERMANOVA run with unrestricted permutation of raw data. Significant results were followed by post hoc tests to determine which mortars at which interval were significantly different. Multidimensional scaling (MDS) plots were used to visually demonstrate assemblage similarity between mortars at each interval. To test for similarity between mortar assemblages, SIMPER analyses were run.

As the mortar samples were not randomised on the crates and kept in their mortar mix groups, a three-ANOVA was run to determine if there were edge effects, with the factor 'location' ('interior' for mortar blocks in the middle of the crate and 'exterior' for mortar blocks on the edge of the crate), 'mortar' and 'interval' tested against all dependent variables.

4.4 Results

It was determined that there were no discernible edge effects arising from the experimental design. Mortar block location on the crate was responsible for 0.2% to 1.1% of explained variance for Hamble Harbour dependent variables. For Poole Harbour, mortar block location was responsible for 0.004% to 0.9% of explained variance, except for count abundance, where mortar block location explained 7.4%, mortar 9%, and interval 18%.

The biota colonising the mortar blocks were typical of shallow subtidal communities (Figure 8), including colonial and solitary sea squirts, barnacles and bryozoans (Table 3). It should be noted that the plastic crates, metal chains and plastic cable ties used in the experimental set up were also significantly fouled at the end of the study period. The undersides of the crates were entirely covered in solitary sea squirts, with the crate sides dominated by similar communities as the blocks though this was not formally quantified. Crabs and small benthic fish were often found in the interstices of the plastic crate. Seven non-native species were recorded in low abundances, with all 7 recorded in Hamble Harbour and 5 recorded in Poole Harbour.

Table 3. Species list with average abundance per block. Standard error (±) given in brackets. Non-native taxa indicated with an asterisk (*). Taxa indicated with a dagger (†) denote those recorded as numeric counts, all other taxa were recorded as percentage cover.

	Poole Harbour			**Hamble Harbour**		
	OPC	**GGBS**	**VP**	**OPC**	**GGBS**	**VP**
Red Seaweeds						
Filamentous	0	0	0	1(0)	0	1(0)
Branching coralline	0	0	0	0	0	1(0)
Green Seaweeds						
Ulva sp.	1(0)	1(0)	1(0)	0	0	1(0)
Porifera						
Grantia compressa†	1.4(0.2)	1(0)	2(0.3)	1.6(0.2)	1.4(0.2)	1.3(0.1)
Halichondria panacea	1(0)	1(0)	1.3(0.3)	11(4.7)	3.2(0.8)	5.8(2.9)
Sycon ciliatum†	0	0	0	0	1.7(0.3)	1.5(0.5)
Ascidians						
Ascidia mentula†	0	0	0	1.1(0.1)	1(0)	1.5(0.5)
Ascidiella aspersa†	2.24(0.4)	2.3(0.3)	3.2(0.4)	1.9(0.2)	2.3(0.3)	2.4(0.2)
Botrylloides leachii	1(0)	0	1(0)	1(0)	1.3(0.3)	2(1)
Botrylloides violaceus	0	0	0	1(0)	0	2.3(1.3)
Botryllus schlosseri	1(0)	1(0)	1(0)	4.6(2.7)	1(0)	5.5(3.0)
Ciona intestinalis†	1.2(0.2)	1.1(0.1)	1.3(0.2)	0	1(0)	1.3(0.3)
*Corella eumyota**†	1.4(0.2)	1(0)	1.1(0.1)	1(0)	1(0)	1.3(0.2)
*Didemnum vexillum**	4.8(2.5)	1.5(0.4)	1.3(0.3)	3.4(2.4)	1.3(0.3)	4.8(2.4)
Molgula sp. †	0	0	0	1.1(0.08)	1.6(0.2)	1.6(0.2)
*Styela clava**†	0	0	0	1(0)	1(0)	1(0)
Bryozoa						
Bugula sp.	0	0	0	3.1(1.4)	7.7(2.6)	5.5(1.6)
Eucratea loricata	1.3(0.1)	1(0)	1(0)	22(2.2)	6(1.2)	21.7(2.5)
Membranipora membranacea	1(0)	2(0)	0	2.3(1.3)	1(0)	1.5(0.3)
*Watersipora subatra**	1(0)	1(0)	1(0)	3.1(1.3)	1.5(0.5)	1.5(0.3)
Polychaetes						
*Ficopomatus enigmaticus**	0	0	0	11.2(2.7)	5.9(1.3)	4.2(0.8)
Polychaete, muddy tubes	3.5(1.3)	2.3(0.4)	1.3(0.2)	1.1(0.08)	1.8(0.7)	1.8(0.5)
Spirobis spirobis	1(0)	1(0)	1(0)	0	1(0)	0
Spirobranchus triqueter	1(0)	1(0)	1.8(0.2)	1(0)	2.3(1.3)	1(0)
Crustacea						
*Amphibalanus amphitrite**	1(0)	0	0	1(0)	1(0)	1(0)
*Austrominius modestus**	1(0)	1(0)	1(0)	1.2(0.2)	1.7(0.3)	6.2(1.7)
Semibalanus balanoides	1(0)	1(0)	1(0)	1(0)	1(0)	1(0)
Mollusca						
Mya arenaria†	0	0	0	0	1(0)	1(0)
Mytilus edulis†	1(0)	1(0)	1(0)	1(0)	1(0)	1(0)
Ostrea edulis†	1(0)	1(0)	1(0)	1(0)	1(0)	1(0)
Total	**20**	**18**	**18**	**24**	**26**	**29**

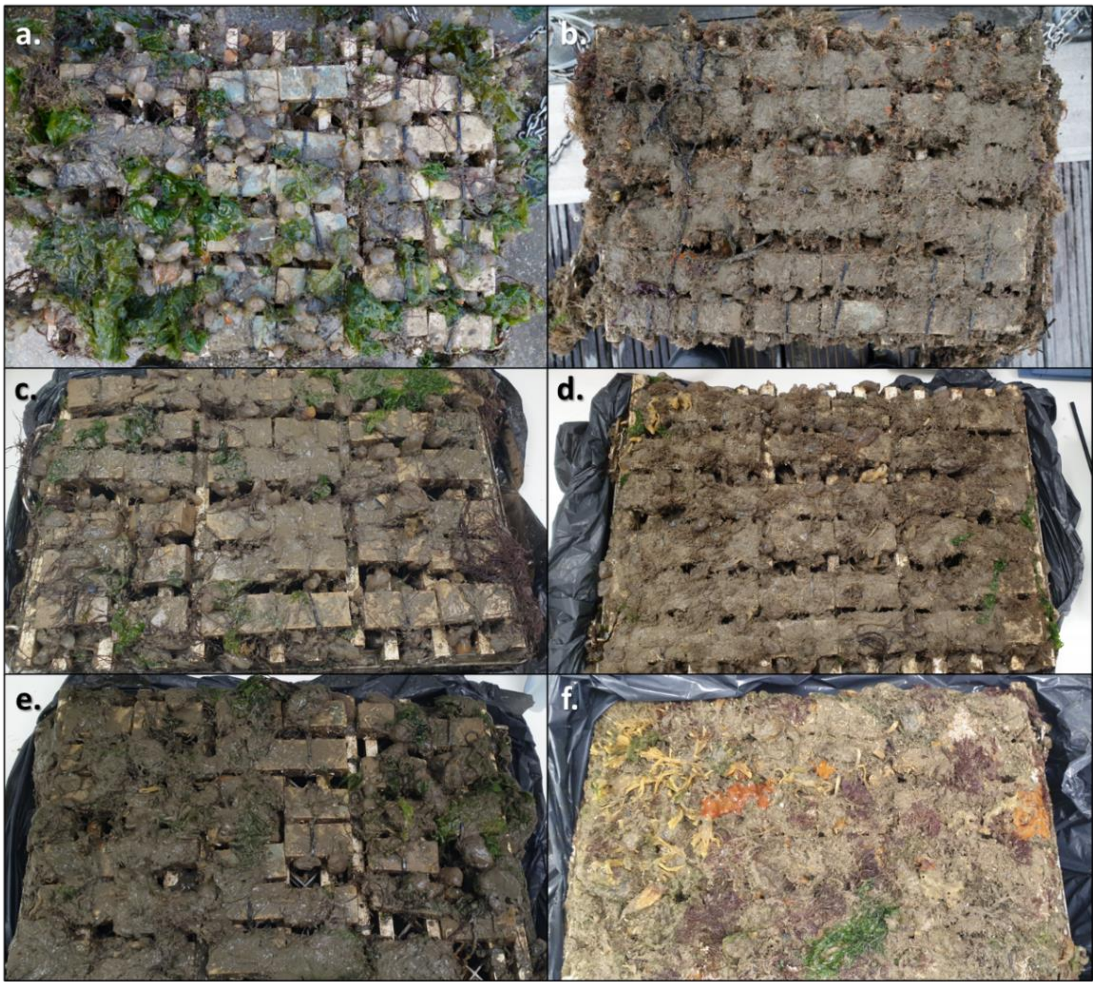


Figure 8. The colonisation of the crates in Poole Harbour (left column) and Hamble Harbour (right column) at 3 months (a, b), 6 months (c, d) and 12 months (e, f).

4.4.1 Species richness

An overall total of twenty and thirty species were recorded over the 12 months on all mortars in Poole Harbour and Hamble Harbour respectively. The highest overall total on a single mortar mix over 12 months in Poole Harbour was 20 species on OPC, and 18 species on both GGBS and VP. In Hamble Harbour, each mortar mix yielded 29, 24, and 26 species on VP, OPC and GGBS respectively over the course of 12 months.

Main tests showed significant interactions for interval and between mortar * interval at Poole Harbour (Table 5), and for all factors at Hamble Harbour (Table 4). Survey interval accounted for the largest proportion (40.72% and 10.5% respectively) of the variance in species richness between factors at Hamble Harbour and Poole Harbour. At both sites, there were no significant differences between the mortar species richness at the 3- or 6-month intervals (Figure 5). Hamble Harbour was slightly more species rich at 3 months than Poole Harbour, but otherwise species richness between the harbours was relatively similar. In both harbours, VP mortar was significantly more species rich than OPC after 12 months, and significantly more species rich than GGBS in Hamble Harbour only.

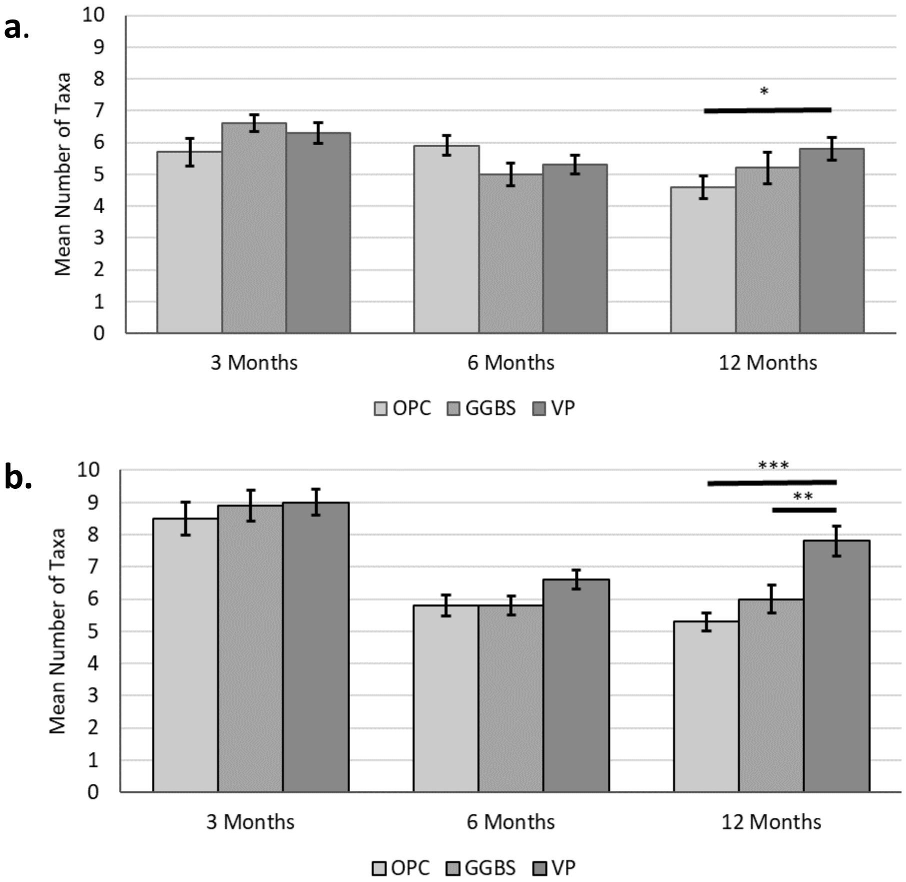


*Figure 9. The mean number of taxa identified on the mortar blocks (OPC – ordinary Portland cement, GGBS – ground granulated blast furnace slag, VP – Vicat Prompt) at 3, 6 and 12 months in Poole Harbour (a) and Hamble Harbour (b). Statistically significant differences indicated by * (<0.05), ** (<0.01) and *** (<0.001). Error bars show standard error.*

Table 4. Main test results for species richness, abundance, and biomass in Hamble Harbour. Bold values indicate significant result.

Factor	**df**	**Deviance**	**Resid. Df**	**Resid. Dev**	***P***	**% Explained**
Species Richness						
Mortar Type	2	4.37	105	50.59	**<.0001**	7.95
Survey Interval	2	22.38	103	28.21	**<.0001**	40.72
Mortar * Interval	4	3.19	99	25.03	**<.0001**	5.80
Abundance - Percentage Cover						
Mortar Type	2	301.51	105	685.54	**<.0001**	30.55
Survey Interval	2	5.82	103	679.72	**<.0001**	0.59
Mortar * Interval	4	122.17	99	557.55	**<.0001**	12.38
Abundance - Counts						
Mortar Type	2	36.45	105	388.73	**<.0001**	8.57
Survey Interval	2	73.11	103	315.63	**<.0001**	17.19
Mortar * Interval	4	10.40	99	305.23	**<.0001**	2.45
Biomass - Loss on Ignition						
Mortar Type	2	1.37	103	61.58	**<.0001**	2.18
Survey Interval	2	44.43	101	17.14	**<.0001**	70.59
Mortar * Interval	4	0.50	97	16.64	**<.0001**	0.79

Table 5. Main test results for species richness, abundance, and biomass in Poole Harbour. Bold values indicate significant result.

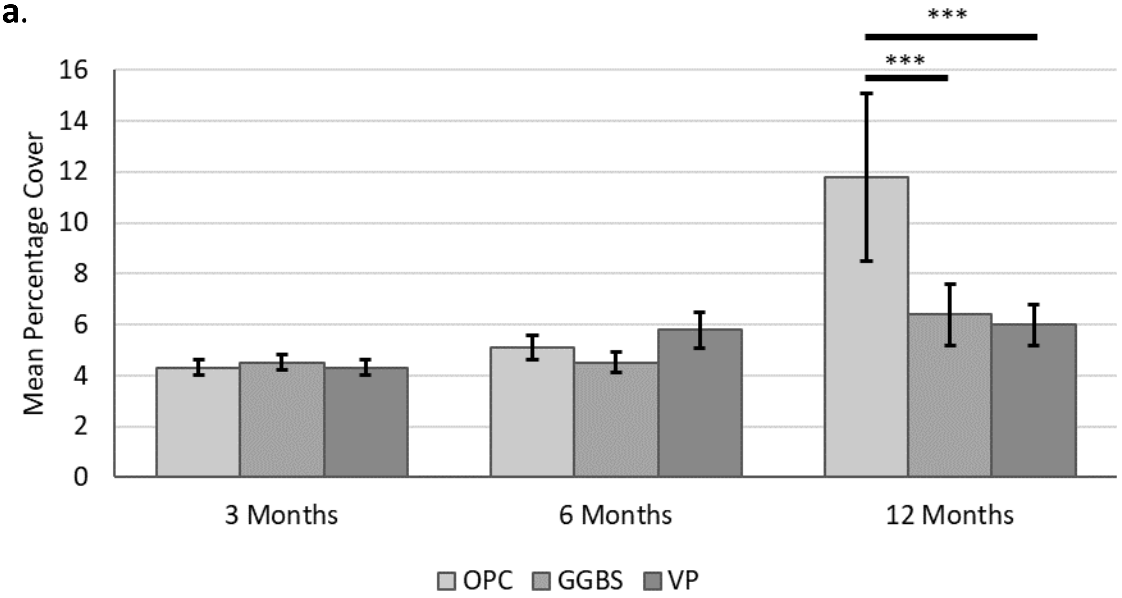
Factor	df	Deviance	Resid. Df	Resid. Dev	*P*	% Explained
Species Richness						
Mortar Type	2	0.5	104	34.9	0.3948	1.5
Survey Interval	2	3.6	102	31.3	**0.0019**	10.2
Mortar * Interval	4	3.1	98	28.2	**0.0317**	8.6
Abundance - Percentage Cover						
Mortar Type	2	13.6	104	242.4	**<0.0001**	5.3
Survey Interval	2	45.8	102	196.6	**<0.0001**	17.9
Mortar * Interval	4	18.1	98	178.5	**<0.0001**	7.1
Abundance - Counts						
Mortar Type	2	11.9	104	165.3	**<0.0001**	6.7
Survey Interval	2	47.7	102	117.6	**<0.0001**	26.9
Mortar * Interval	4	5.5	98	112.1	**<0.0001**	3.1
Biomass - Loss on Ignition						
Mortar Type	2	1.3	102	33.3	0.1369	3.7
Survey Interval	2	1.4	100	31.8	0.1104	4.1
Mortar * Interval	4	2.5	96	29.4	0.1087	7.1

4.4.2 Abundance

Unlike species richness, percentage cover trends of biota differ between harbours. In Poole Harbour, percentage cover on a single block does not exceed 32% at any one time and, on average, remains relatively low. Main tests show significant interactions for all factors, with survey interval accounting for the largest proportion (17.9%) of the variance in percentage cover (Table 4). There are no significant differences between the percentage cover of mortars at 3- and 6-months, but at 12-months percentage cover on OPC is significantly higher than both GGBS and VP (Figure 10).

In Hamble Harbour, percentage cover is generally much higher, and varies between mortars at each survey interval. Main tests showed significant results for all factors, with mortar type accounting for the largest proportion (30.55%) of the variance in percentage cover (Table 5). At both 3- and 6-month intervals, percentage cover on OPC and VP was significantly higher than GGBS. At 12-months, percentage cover on OPC significantly higher than both GGBS and VP. Despite the magnitude of percentage cover difference between both harbours, at 12-months the distribution of percentage cover between the mortars is very similar.

a.



b.

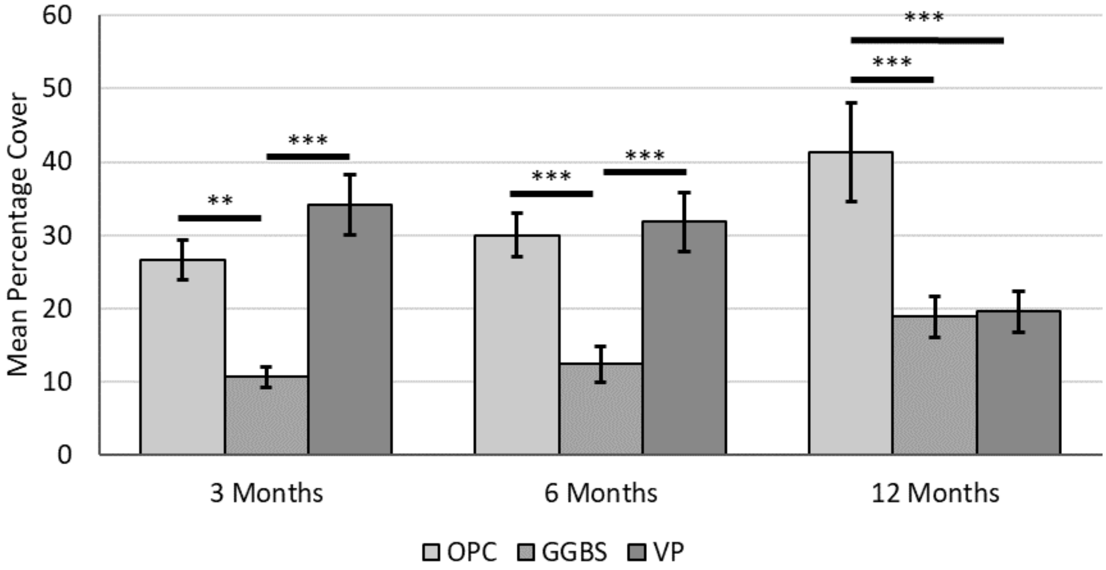


*Figure 10. The mean percentage cover on the mortar blocks (OPC – ordinary Portland cement, GGBS – ground granulated blast furnace slag, VP – Vicat Prompt) at 3, 6 and 12 months in Poole Harbour (a) and Hamble Harbour (b). Statistically significant differences indicated by * (<0.05), ** (<0.01) and *** (<0.001). Error bars show standard error.*

In Poole Harbour, numeric counts of abundance decrease over time for all mortar formulas and Poole Harbour was generally less abundant again than Hamble Harbour. Main tests showed significant results for all factors, with survey interval accounting for the largest proportion (26.9%) of the variance in numeric counts (Table 4). Numeric counts for OPC were significantly lower than both GGBS and VP at the 3-month interval. At the 6-month interval, there was no significant differences between the numeric counts of mortar formulas, but at the 12-month interval numeric counts on VP were significantly higher than GGBS (Figure 11).

Main tests showed significant interactions for all factors Hamble Harbour (Table 5), with survey interval again accounting for the largest proportion (17.19%) of the variance. At 3- and 6-month intervals, numeric counts were relatively similar between mortars and intervals, but were significantly less on OPC than both GGBS and VP at the 12-month interval (Figure 11).

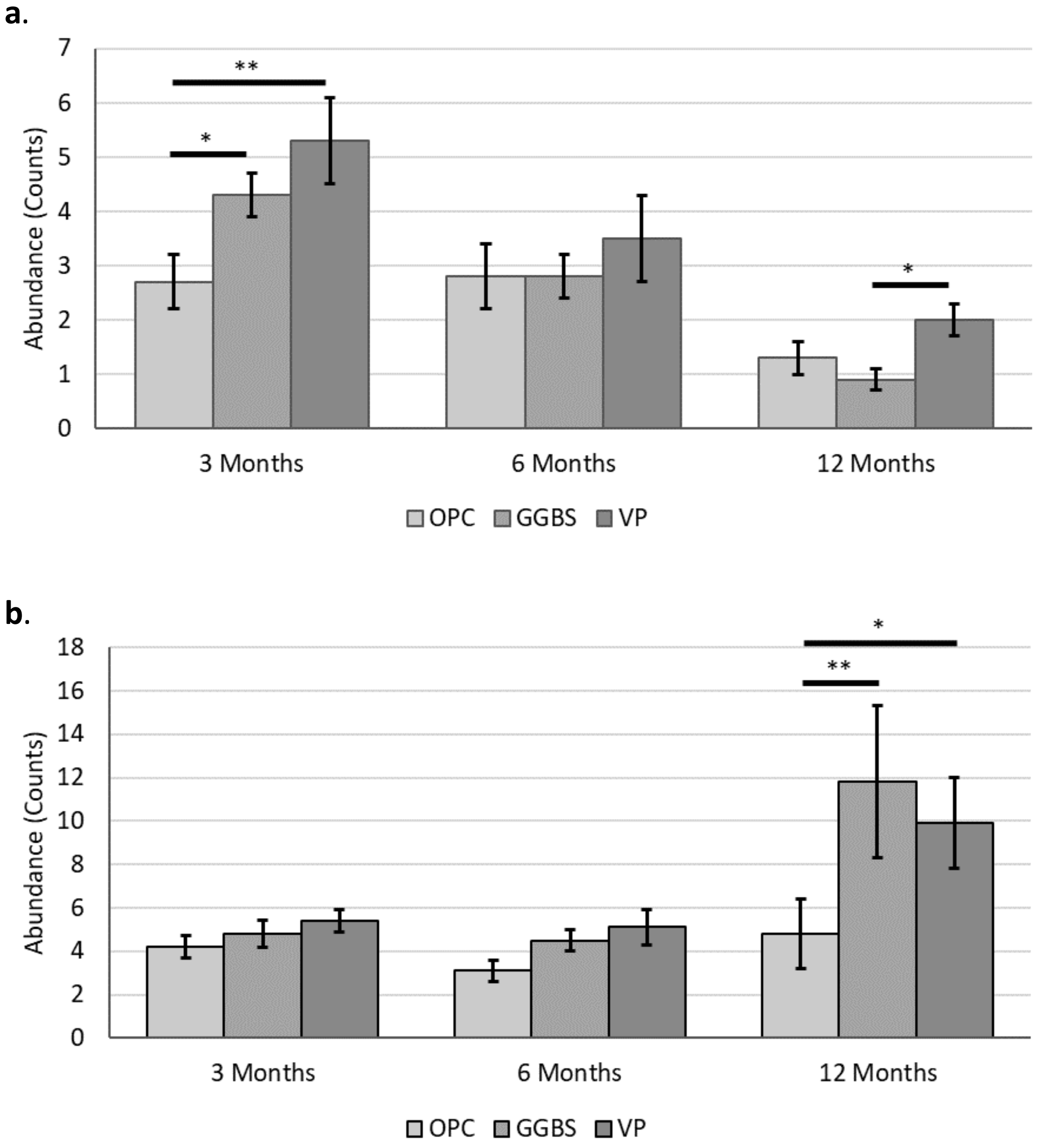


*Figure 11. The mean abundance (counts) on the mortar blocks (OPC – ordinary Portland cement, GGBS – ground granulated blast furnace slag, VP – Vicat Prompt) at 3, 6 and 12 months in Poole Harbour (a) and Hamble Harbour (b). Statistically significant differences indicated by * (<0.05), ** (<0.01) and *** (<0.001). Error bars show standard error.*

4.4.3 Biomass

For both harbours and all mortars, mean loss on ignition (LOI) does not exceed 0.8 g for 3- and 6-month intervals. Main tests for Poole Harbour showed no significant interactions for mortar, interval or mortar*interval (Table 4). In Poole Harbour, biomass on OPC blocks is significantly greater than GGBS at 12 months (Figure 12). Biomass for GGBS and VP at 12 months remains similar to biomass recorded in previous intervals. In Hamble Harbour, main tests showed significant interactions for all factors

(Table 5) with survey interval accounting for the largest proportion (70.59%) of the variance, though there were no significant differences between mortars at any survey interval (Figure 12). Biomass for all mortars in Hamble Harbour at the 12-month interval is greater than previous survey intervals.

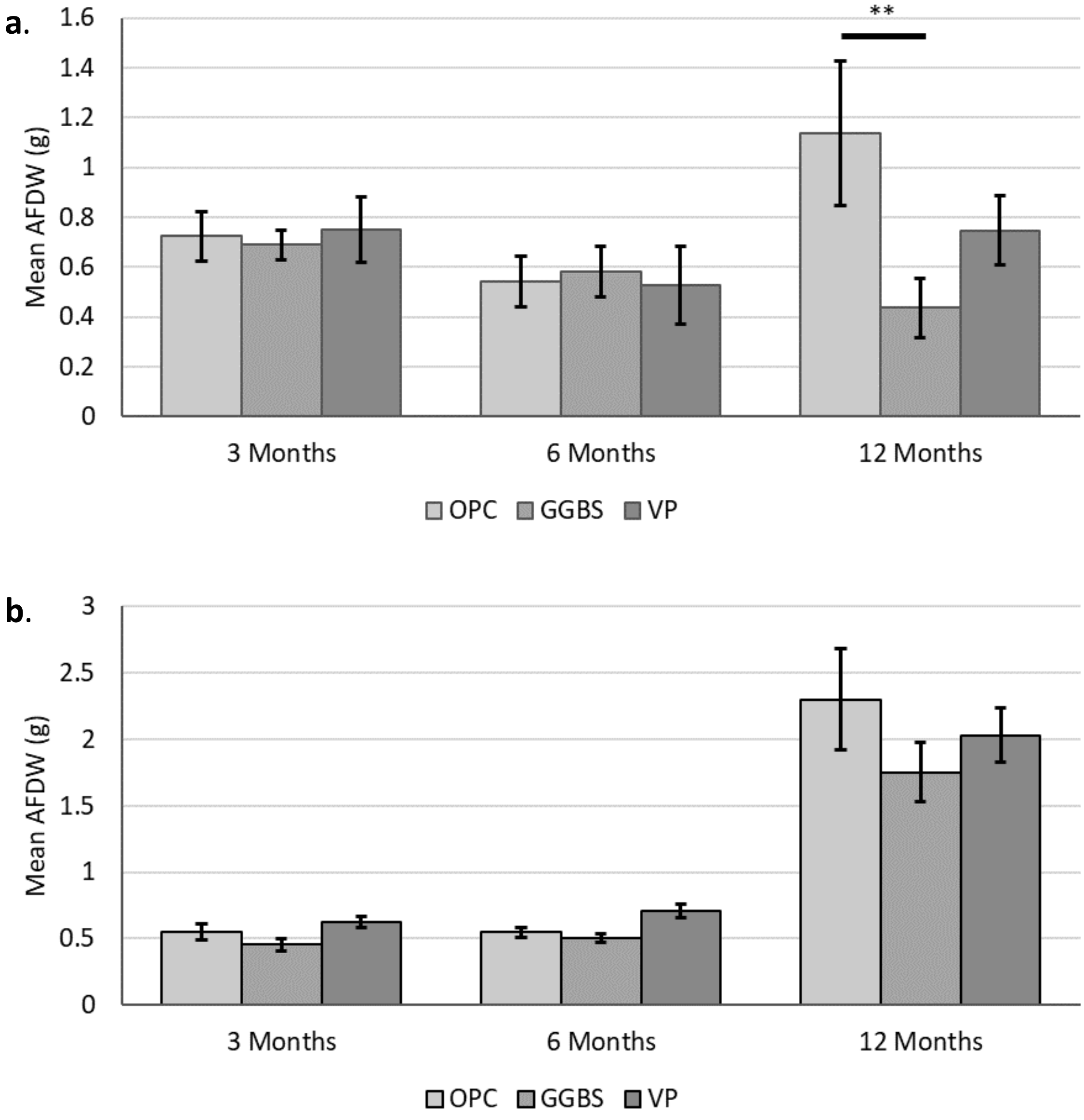


*Figure 12. The mean loss on ignition on the mortar blocks (OPC – ordinary Portland cement, GGBS – ground granulated blast furnace slag, VP – Vicat Prompt) at 3, 6 and 12 months in Poole Harbour (a) and Hamble Harbour (b). Statistically significant differences indicated by * (<0.05), ** (<0.01) and *** (<0.001). Error bars show standard error. Note difference in Y-axis scales between a) and b).*

4.4.4 Assemblages

SIMPER analysis showed the average dissimilarity between the two harbours was 77%, with the average similarity within Poole Harbour 45% and the average similarity with Hamble Harbour 42%. The Poole Harbour assemblages were dominated by solitary ascidians and low abundances of other sessile taxa, but Hamble Harbour, although dominated by bryozoans, had comparatively greater abundances of other taxa, particularly calcareous tubeworms and barnacles.

At Hamble Harbour, average similarity within mortar blocks ranged from 41% to 45%, with the same three species (*Ficopomatus engimaticus, Eucratea loricata, Ascidiella aspersa*) contributing at least 60% of the assemblage similarity for each mortar. The average dissimilarity between mortar blocks ranged from 57% to 60%, with OPC & VP showing the least dissimilarity and OPC & GGBS showing the greatest dissimilarity. A PERMANOVA test indicated significant differences in assemblage structure between mortars, survey interval and mortar*interval (Table 6). Post hoc tests showed that at 3 months GGBS was significantly different to OPC and VP, at 6 months all the mortars were significantly different from each other, and at 12 months only OPC and VP were significantly different to each other (Table S7, Supplementary Material). This is reflected in multidimensional scaling plots for 3-, 6- and 12-month intervals (Figure 13).

At Poole Harbour, average similarity within mortar blocks ranged from 43% to 48%, with the same three species (*Spirobranchus triqueter, Ascidiella aspersa,* polychaete) contributing at least 70% of the assemblage similarity for each mortar. The average dissimilarity between mortar blocks ranged from 54% to 56%, with GGBS & VP showing the least dissimilarity and OPC & VP showing the greatest dissimilarity. A PERMANOVA test indicated significant differences in assemblage structure between mortars, survey interval and mortar*interval (Table 6). Post hoc tests (Table S8, Supplementary Material) showed that at 3 months all the mortars were significantly different from each other, at 6 months GGBS was significantly different to both OPC and VP, and at 12 months only OPC and VP were significantly different to each other (Figure 13).

In both harbours, the mortar blocks follow broadly similar patterns in changes in assemblage structure; initially (3 to 6 months), the mortar blocks are significantly different from each other with this appearing more consistent with GGBS. However, at 12 months in both sites, GGBS is not significantly different to the other mortars, but OPC * VP are significantly different to each other.

*Table 6. Individual PERMANOVA results for tests between mortar, interval and mortar*interval on species abundance data (number of permutations 9999).*

Source	**df**	**SS**	**MS**	**Pseudo-F**	***P* (perm)**
Hamble Harbour					
Mortar	2	8820.2	4410.1	5.4275	**0.0001**
Interval	2	73020	36510	44.933	**0.0001**
Mortar * Interval	4	8608.6	2152.2	2.6487	**0.0016**
Poole Harbour					
Mortar	2	5417.2	2708.6	3.2134	**0.0002**
Interval	2	30898	15449	18.328	**0.0001**
Mortar * Interval	4	7851.1	1962.8	2.3285	**0.0004**

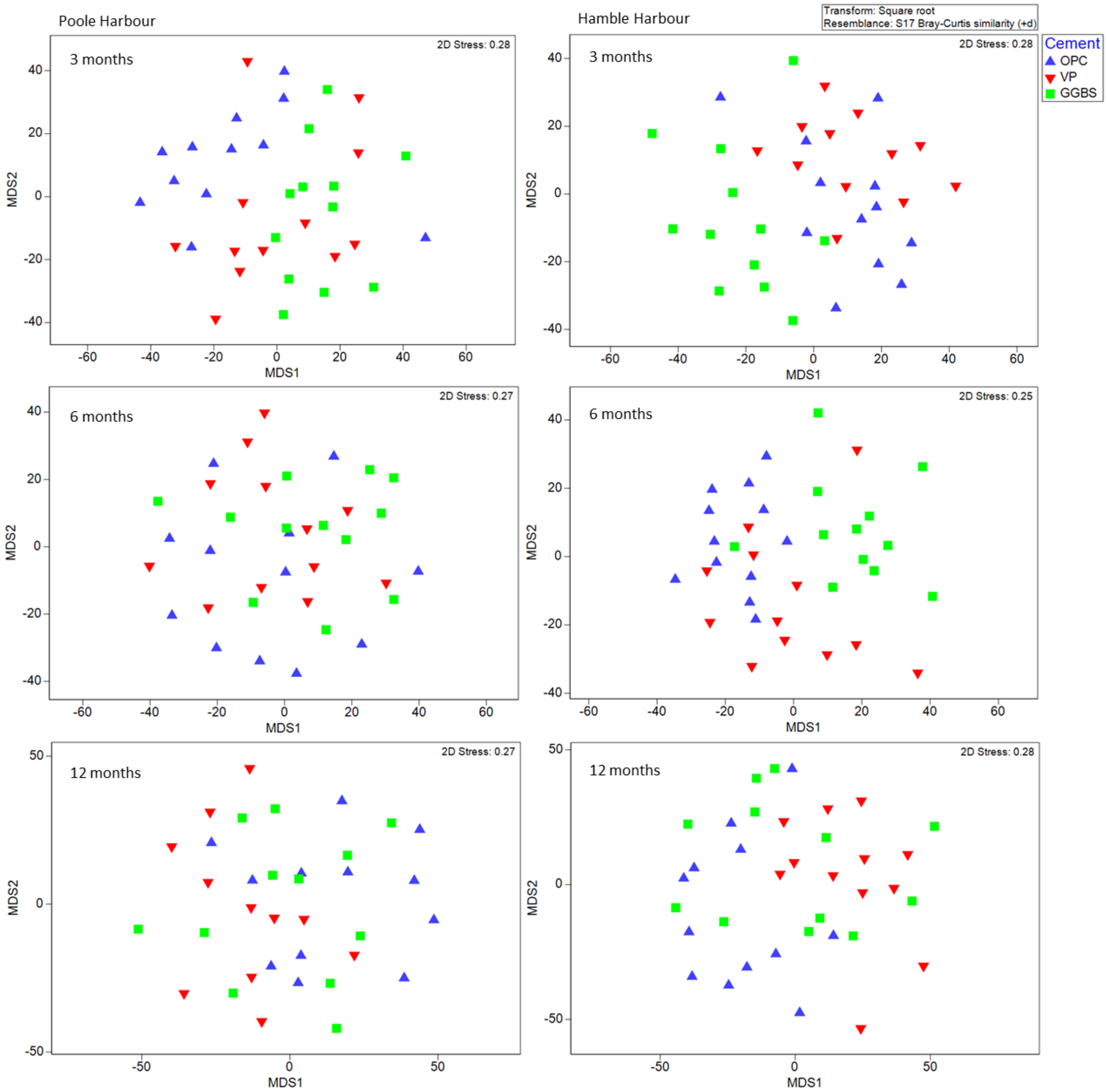


Figure 13. Multidimensional scaling plots indicating Poole Harbour (left) and Hamble Harbour (right), with 3 months (top), 6 months (middle) and 12 months (bottom), using species abundance data. Blue triangles indicate ordinaryPortland cement, red triangles indicate Vicat Prompt and green squares indicate ground granulated blast furnace slag assemblages.

4.5 Discussion

In this study, the bioreceptivity of two low carbon cements VP (Vicat Prompt) and GGBS (ground granulated blast furnace slag) were compared to OPC (ordinary Portland cement). With the exception of percentage cover at 3- and 6-month intervals in Hamble Harbour, and counts after 3 months in Poole Harbour, there are no significant differences between mortars at 3 and 6 months for species richness, abundance or biomass. After 12 months, VP is significantly more species rich than both OPC and GGBS in Hamble Harbour and OPC in Poole Harbour, and significantly more abundant (numeric counts only) than GGBS in Poole Harbour and OPC in Hamble Harbour. However, the control (OPC) mortar shows significantly higher percentage cover than GGBS and VP in both harbours, and higher biomass in both harbours than VP though this was not significant. After 12 months, the assemblage structure of VP blocks is significantly different to OPC.

These results suggest that after 12 months VP may offer some enhanced bioreceptivity compared to OPC and GGBS mortars, but this is not consistent between harbours or metrics. Equally, in circumstances where VP does not show enhanced bioreceptivity (biomass, percentage cover), the control mortar using OPC demonstrates greater bioreceptivity. The bioreceptivity of VP after 12 months in this study is variable and therefore its use in concrete structures to enhance the primary chemical bioreceptivity may not perform as desired. It is recommended that its use is accompanied by other factors that are known to enhance bioreceptivity, such as varying the surface texture (MacArthur et al. 2019; Hayek et al. 2021; Bone et al. 2022).

The differences in the data between 6 and 12 months suggests that succession may be ongoing, particularly as climax communities on coastal structures can take between 5 and 20 years to form (Hawkins et al. 1983; Pinn et al. 2005; Coombes 2011). Further study would benefit from surveying replicates annually for at least five years. The colonisation and succession of the mortar blocks in this study may have been hindered by the timing of deployment (June 2021, boreal summer), having missed peak settlement season for many sessile species, though this may not impact colonisation outcomes longer term (Naylor et al. 2023). Exogenous factors, such as larval supply (Strain et al. 2021), can impact the bioreceptivity of concrete (Bone et al. 2022). This could explain some of the differences seen between 6 (December 2021, boreal winter) and 12 months (June 2022, boreal summer) that were not seen between 3 (September 2021, boreal autumn) and 6 months. The different assemblage structures observed in the two harbours further supports this.

Guillitte and Dreesen (1995) found the polystyrene rests that housed their test materials were also colonised by the photosynthetic microorganisms used in their experiment. They suggested that the colonising organisms were less dependent on the substrate bioreceptivity, but more ambient exogenous factors such as nutrient availability. In our study, the crates were heavily fouled by assemblages that appeared to be very similar to those on the mortar blocks. The set-up materials (plastic, stainless steel) had smooth surfaces akin to the mortar blocks, and the cm-scale holes, ledges and gaps were well utilised by solitary ascidians and small mobile fauna. This observation suggests that substrate material may be less important than overall structural complexity.

GGBS-based concrete, in the initial stages of colonisation, is reported to be superior to 100% OPC due to the lower alkalinity (Morin et al. 2018; Hayek et al. 2020; 2021) though these lab-based studies focussed on microorganisms and lasted no longer than 28 days. The results of this current study do not reflect these findings, with GGBS only showing significantly greater abundance (numeric counts) at 3 months in Poole Harbour and 12 months in Hamble Harbour. Otherwise, biomass, percentage cover and species richness are either similar or significantly less and vary between sites. As no data were collected prior to 3 months, it is not possible to say whether early bioreceptivity was greater on GGBS mortar blocks. Other studies have shown that concrete containing GGBS is not always bioreceptively superior to OPC (McManus et al. 2018). When in different environments in the field, GGBS and OPC have been shown to vary in their bioreceptive performance. Natanzi et al. (2021) found that OPC/GGBS concrete blend had greater biomass than OPC only concrete tiles on the sheltered side of a breakwater in Ireland, but there was no difference on the exposed side. Additionally, when surface texture is factored in it can dominate any effect of cement chemistry on bioreceptivity (Hayek et al. 2021).

Natural (single source) cements are not as readily available as artificial (blended) OPC, and Vicat produce the only prompt natural cement at an industrial scale (Baxter et al. 2022). In addition to its rapid setting, the limited source and relatively high market cost likely make it unsuitable for large scale application, particularly given the cost-driven barriers to ecological enhancements in coastal construction and engineering (Kleijn et al. 2019; Sella et al. 2022). However, VP's rapid setting makes it suitable in the application of smaller retrofit ecological enhancements. GGBS-based concrete is already often used in coastal and marine infrastructure (British Standards Institute 2011) as it reduces the ingress of aggressive substances including water and chlorides (Smith 2016) and increases resistance to sulphate attack and the alkali-silica reaction (Neville 2011). Therefore, budget allocation should be prioritised toward producing micro- and macro-scale surface complexity, which may be achieved in a variety of ways (*sensu* Naylor et al. 2017; Strain et al. 2018; O'Shaughnessy et al. 2019; Evans et al. 2021).

4.6 Conclusion

It is evident from our work that the primary chemical bioreceptivity of OPC, GGBS and VP is inconsistent between metrics and sites. Other intrinsic factors, such as surface roughness and macroscale structural complexity, and exogenous factors, such as local environmental conditions and larval supply, should be prioritised and considered carefully when designing bioreceptivity into ecological enhancements. The concrete used for ecological enhancements should meet minimum industry standards and prioritise sustainability through low carbon binders and recycled aggregates, and longevity and durability.

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4.8 Supplementary material

Table S7. PERMANOVA post hoc test results for Hamble Harbour. Bold values indicate significant result.

Groups	t	*P* (perm)	Unique perms
3 Months			
OPC * VP	1.2877	0.1173	9909
OPC * GGBS	2.0146	**0.0003**	9909
VP * GGBS	2.3704	**0.0001**	9902
6 Months			
OPC * VP	2.0499	**0.0003**	9919
OPC * GGBS	2.8014	**0.0001**	9915
VP * GGBS	2.1265	**0.0001**	9918
12 Months			
OPC * VP	2.1245	**0.0001**	9891
OPC * GGBS	1.3216	0.0972	9906
VP * GGBS	1.2126	0.1729	9895

Table S8. PERMANOVA post hoc test results for Poole Harbour. Bold values indicate significant result.

Groups	t	*P* (perm)	Unique perms
3 Months			
OPC * GGBS	2.1683	**0.0003**	9928
OPC * VP	1.5705	**0.046**	9922
GGBS * VP	1.7628	**0.013**	9925
6 Months			
OPC * GGBS	1.7716	**0.0087**	9913
OPC * VP	1.3218	0.1387	9901
GGBS * VP	1.7881	**0.0093**	9907
12 Months			
OPC * GGBS	0.81982	0.6661	9926
OPC * VP	1.8267	**0.0043**	9901
GGBS * VP	1.4045	0.0851	9922

Chapter 5 – Artificial rockpools create habitat refugia on seawalls at high tide

This chapter is currently being reviewed for publication.

5.1 Abstract

Rockpools are fundamental habitats on natural rocky shores that provide refugia for marine life at low and high tide. Yet, artificial coastal structures lack the topographical complexity seen in these natural habitats. Eco-engineering, that may include the deployment of artificial rockpools, attempts to address the lack of suitable habitat on coastal infrastructure but most studies focus on species abundance metrics at low tide. It is important to understand how eco-engineering interventions may provide habitat at high tide compared to the surrounding artificial substrate.

In this study, we demonstrate how groups of rockpools (1, 3 or 5 rockpools) add habitat complexity at high tide to a concrete seawall in Poole Harbour, UK. Between April and October 2022, eighteen GoPro cameras were deployed to record species richness and abundance in artificial rockpools and the adjacent concrete sea wall. Additionally, the length of time the most abundant fauna (the shanny fish *Lipophrys pholis* and the European shore crab *Carcinus maenas*) spent engaging in different behaviour (resting, feeding, moving) was recorded.

Overall, there was no significant difference in the abundance and species richness of mobile fauna using the artificial rockpools versus the seawall. However, both the shanny and shore crab generally spent more time in the rockpools than on the seawall. Both crabs and shanny preferentially engaged in feeding behaviours on the seawall, which has been attributed to the high percentage cover of barnacle prey. Crabs and shanny spent more time resting in the rockpools than the seawall and the only reproductive behaviour observed occurred within the rockpools. Our work suggests that artificial rockpools support the habitat needs and multiple life history requirements of these species at high tide. Both the seawall and rockpools provide valuable resources, which further emphasises the need for variety in eco-engineering feature designs.

5.2 Introduction

The intertidal zone interfaces the land and sea, and emersion of the intertidal at low tide exposes fauna to risk of mortality through desiccation stress and predation (Little et al. 2009). Species richness on intertidal rocky shores can be largely attributed to the heterogenous nature of the substratum, which can include rough and smooth surfaces, pools, overhangs and a variety of aspects and steepness of slope. Rockpools are shallow water-retaining depressions (Metaxas and Scheibling 1993) that are found on intertidal rocky shores that provide refuge and resources at low and high tide (Martins et al. 2007; Firth et al. 2013; Legrand et al. 2018). The patchiness of rockpools and other microhabitats are crucial to the connectivity and survival of mobile fauna (Noel et al. 2009) on rocky shores.

Artificial coastal structures, such as seawalls, piers, groynes and breakwaters, often lack the structural and topographical complexity seen in natural rocky shores (Moschella et al. 2005; Chapman and Underwood 2011; Aguilera et al. 2014). As a result, they usually fail to support diverse intertidal benthic communities, and often have lower abundance, species richness and biodiversity (Connell and Glasby 1999; Chapman 2003; Moschella et al. 2005; McKinney 2006; Glasby et al. 2007; Vaselli et al. 2008; Pister 2009; Firth et al. 2013; Earp et al. 2023). Artificial coastal structures provide poorer attachment capabilities for seaweeds (Drakard et al. 2021) and alter reproductive states and outputs of gastropods (Moreira et al. 2006; Thompson et al. 2023) compared to natural substrates.

Intertidal eco-engineering involves the integration of habitat features in construction (Bergen et al. 2001; Mitsch and Jorgensen 2003; Odum and Odum 2003) and can increase benthic species richness on artificial coastal structures (Firth et al. 2014; Naylor et al. 2017; Strain et al. 2017; O'Shaughnessy et al. 2019; Evans et al. 2020). However, the majority of studies are conducted at low tide and only capture data for half of the story. Understanding how eco-engineered habitats are used at high tide is crucial for ensuring the habitat needs of organisms are met throughout the tidal cycle and for further demonstrating how interventions may improve ecosystem provision and functionality of artificial coastal structures (Bishop et al. 2022).

Research on mobile fauna associated with artificial structures has predominantly concentrated on offshore subtidal natural and artificial reefs (Baine 2001; Reis et al. 2021), with a recent focus on industrial subtidal structures, such as renewable energy installations (Wilhelmsson et al. 2006; Reubens et al. 2013; Krone et al. 2013; Reubens et al. 2014; Langhamer et al. 2016; Bender et al. 2020; Glarou et al. 2020; Sheehan et al. 2020) and oil rigs (Claisse et al. 2014; 2015). Investigations into the behaviour of intertidal mobile species have been limited by the availability of suitable equipment. Yet the advent of cheaper, robust underwater cameras has increased the possibilities of simultaneously comparing assemblages of different habitats. For example, using GoPro cameras, Ng et al. (2021) found that the feeding rate of algal turf-feeding fishes was greater on granite seawalls than adjacent reef flats on the coast of Singapore, and the functional composition of algal turf-feeding fishes also differed between seawall and reef habitats. The authors postulated that this was due to the greater algal turf coverage on the seawall habitat.

Using cameras, intertidal ecological enhancements have been shown to have varying impacts when surveying fish and crab abundance, species richness and assemblage composition at high tide (Sheehan et al. 2010; Morris et al. 2017; 2018; Ushiama et al. 2019; Strain et al. 2020; Taira et al. 2020). Results are generally species poor compared to studies conducted at low tide surveying epilithic and sessile biota on ecological enhancements. Morris et al. (2017) found no consistent effect of artificial rockpools (flowerpots) on pelagic and benthic fish assemblages and few effects on diversity and abundance over the year of their study in Sydney Harbour, Australia. Ushiama et al. (2019) found similar results, with no significant difference in pelagic and cryptobenthic fish abundance between complex tiles, flat tiles, and seawall in Sydney Harbour. However, the length of time cryptobenthic fish spent interacting with complex tiles was greater than flat tiles and the seawall, though this difference was not apparent for pelagic fish. Strain et al. (2020) found species density and maximum abundance (MaxN) of cryptobenthic fish was positively influenced by tiles seeded with oysters in Sydney Harbour, though the complexity of the tiles had little effect. Again, there were no detectable effects in the species diversity or MaxN of pelagic fish between seeded and unseeded, and complex and flat tiles after 1 month. Conversely, Taira et al. (2020) found that enhanced granite riprap in Singapore significantly increased species richness of epibenthic fish, as well as influencing their assemblage composition, likely due to the increase in trophic food provision through enhanced algal growth.

As part of the Marineff Project, forty-five artificial rockpools were installed on an intertidal seawall in Poole Harbour, UK. Experimental data collected from the rockpools at low tide demonstrated their success at increasing the species richness of the overall seawall, as well as increasing the abundance of mobile fauna such as crabs, prawns and benthic fish (unpublished data). At the time of writing, all existing studies using video analysis of eco-engineering interventions at high tide have been conducted in warm temperate or tropical locations in the southern hemisphere, and there is a need to investigate the effects of ecological interventions in colder temperate waters in the northern hemisphere. To determine how the artificial rockpools affected local benthic and demersal mobile species compared to the seawall at high tide, video footage of both the artificial rockpools and sections of seawall was recorded. To our knowledge, this is the first study of its kind that examines the individual behaviours of mobile fauna interacting with artificial rockpools at high tide. We hypothesised that:

- As a structurally complex feature, abundance of fauna would be greater in the artificial rockpools, than on seawall controls.
- As an ecologically complex feature, organisms would interact with the artificial rockpools for longer than the seawall controls.
- Organisms would interact more with groups of artificial rockpools, which add more local complexity and potential habitat connectivity, than single rockpools.

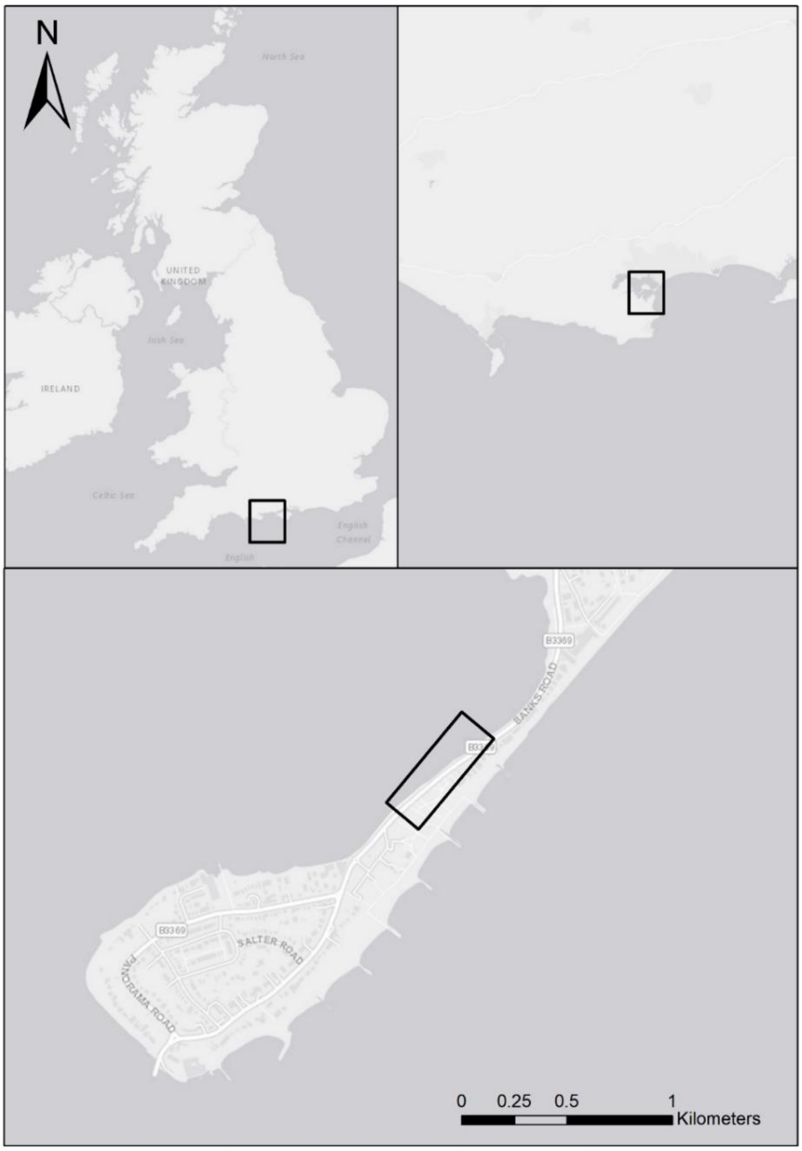
5.3 Method and materials

5.3.1 Location

References to seasons hereafter will refer to boreal seasons. The study site is in Poole Harbour (50.691798, -1.9353187) Dorset, UK (Figure 14). Poole Harbour is a microtidal estuary with a double high tide, where for ~16 hours a day the water is above mean tide level (Humphreys 2005). Salinity ranges between 26.3 ppt to 34.5 ppt in this area of the harbour (Humphreys 2005). The Peninsula, known as Sandbanks, is highly developed and a very popular area for tourism and water sports activities, particularly during the summer.

The study site was on a north-west facing vertical concrete seawall which was characterised by 75% mean percentage cover of barnacles and 20% mean percentage cover of fucoid algae. The site consisted of ~180 m of contiguous seawall, which included an 85 m stretch of seawall between two concrete boat slipways on which forty-five retrofitted artificial rockpools were fixed, and an additional 60 m of seawall the other side of one of the slipways where 6 of the control replicates were installed. The additional seawall was necessary as not all control camera replicates would fit between rockpool treatments.

Artificial rockpools were installed at High Water Neap tidal level in July 2020. The rockpools were made from low-carbon Vicat Prompt cement and sharp sand by Artecology Ltd. and fixed to the seawall using marine-grade stainless steel brackets (Figure 15). The pool interior and rim had a rough, stippled texture and the exterior was covered in concave hemispherical pockets created using bubble wrap to line the rockpool moulds. The rockpools were arranged in three different treatments: five rows of five, five rows of three and five single rockpools. In each row, the rockpools were separated by 15 cm of seawall (Figure 15) and were fully immersed when the tidal height exceeded ~2 m (high water springs maximum 2.5 m) and retained approximately 1.5 litres of seawater at low tide. At the commencement of the experiment in April 2022, the rockpools were characterised by 60% mean percentage cover of fucoids and 6% mean percentage cover of barnacles. Species on the rockpools and seawall included the brown alga *Fucus spiralis*, green alga *Ulva* spp., red turf algae, littorinid snails, patellid limpets, the native barnacle *Semibalanus balanoides* and the non-native barnacle *Austrominius modestus*.



Service Layer Credits: Esri, HERE, Garmin, (c) OpenStreetMap contributors, and the GIS user community

Figure 14. The location of the study site in Poole Harbour on the south coast of the UK.

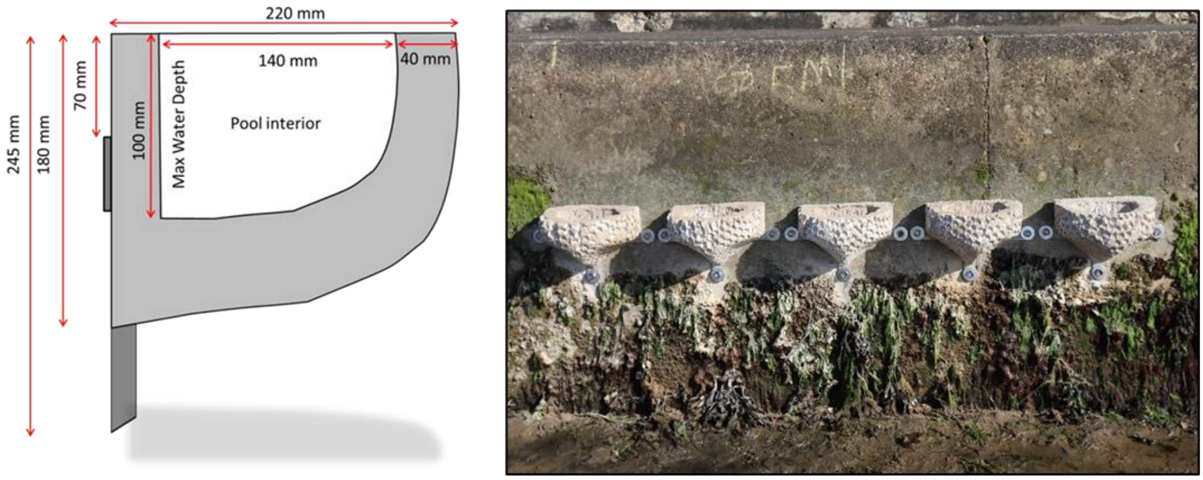


Figure 15. The profile view of an artificial rockpool showing approximate dimensions (left) and a group of 5 rockpools at low tide shortly after installation in July 2020 (right).

5.3.2 Experimental design

Three replicates of each rockpool treatment (groups of five, groups of three, single rockpools) were randomly selected for filming. For the grouped rockpools, only the central rockpool was filmed with the assumption that this position would be representative of species activity and abundance within each pool of the array (Figure 16). In addition to the nine rockpools being filmed, nine sections of the seawall without rockpools were also filmed as controls. Each rockpool treatment and control section was at least >2 m from the next camera to help maintain independence and avoid spatial autocorrelation. Morris (2016) showed that little spatial correlation in the diversity or abundance of fish between adjacent cameras on seawalls occurs.

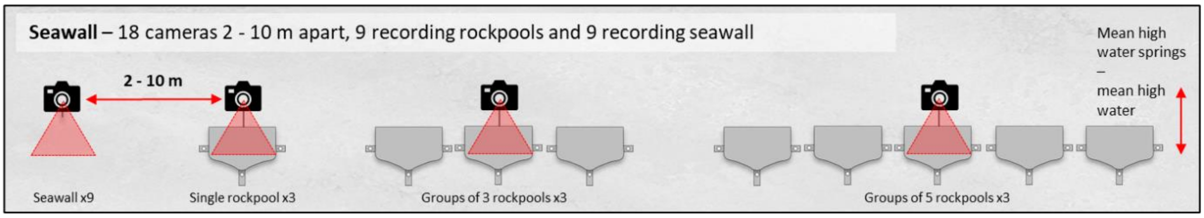


Figure 16. Experimental set up (not to scale).

In each of the nine rockpools, a 10 mm diameter hole was drilled into the rim of the rockpool, adjacent to the seawall. A small amount of resin mortar was injected into the hole and a 24 mm stainless steel connector (or coupling) nut was set into the resin mortar. This was used as a point of attachment for a stainless-steel threaded bar which was screwed into the nut, and a tripod mount screwed onto the other end. A GoPro Hero9 camera in a dive housing was fixed to the tripod mount (1080 progressive resolution, 24 frames per second, wide lens, vertical field of view 94.4º, horizontal field of view 122.6º). The camera was then angled to face down at the top elevation of the rockpool (Figure 17, Morris et al. 2017; Strain et al. 2020). This was replicated for the seawall, where the holes were drilled into the concrete facing, ensuring the same distance from the sea wall. The control cameras on the seawall were fixed and angled to capture the same tidal height as the cameras above the rockpools (Figure 17). No bait was used to prevent bias in the organisms attracted to the cameras (Whitmarsh et al. 2017), particularly as there were a high number of camera replicates on the same structure, and so bait plumes may have overlapped into neighbouring habitats (Hannah and Blume 2012) and disrupted independence. Additionally, unbaited remote underwater video is sufficient at capturing the intertidal species likely to appear at this estuarine site (Rhodes et al. 2020). For the rockpool cameras, the rockpool rim was within the camera frame. To ensure an equivalent and comparable spatial area an equivalent tidal depth was recorded on the seawall, the seawall was marked with a thin line of paint to prevent recording of organisms beyond the limit of the rockpool depth.

From April to October 2022, continuous footage was recorded simultaneously on all replicates (total 18 cameras) for two diurnal high tides each month during late morning. Where possible, these were consecutive diurnal high tides. Footage began recording on a flooding tide as the cameras were immersed (Becker et al. 2012; Espadero et al. 2020) at 2.1m above Chart Datum and was recorded for for 60 minutes (Whitmarsh et al. 2017). Due to the microtidal regime of the harbour, it was often not possible to record more than 60 minutes of immersed footage. For one tide in July and both tides in August and September only two replicates of the single rockpools were filmed as a nut broke away from one of the single rockpools.

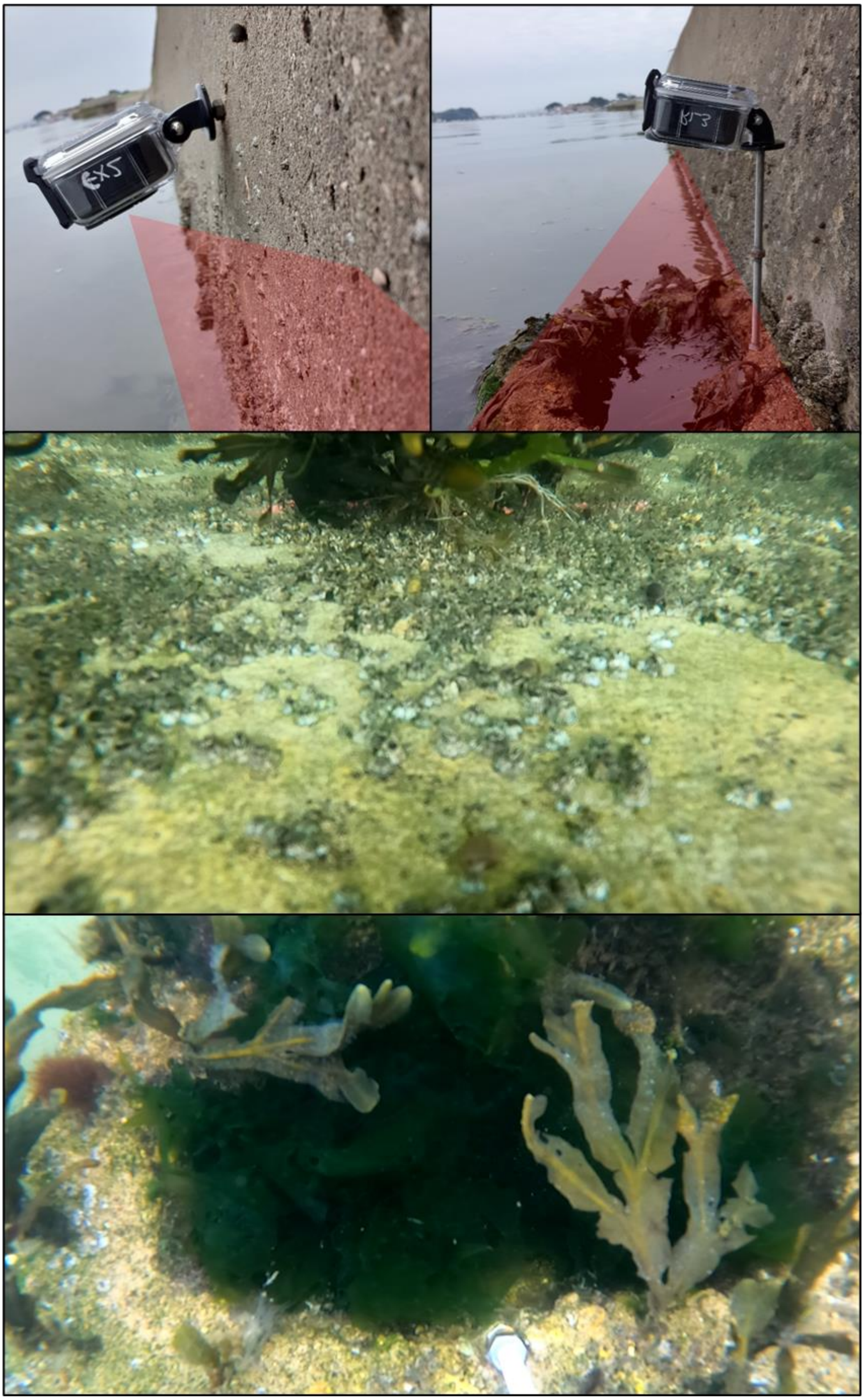


Figure 17. The installation of the GoPro Hero 9s above the seawall (top left) and above the rockpools (top right) with the red shaded area showing the field of view, and the view underwater of the seawall cameras (middle) and the rockpool cameras (bottom).

5.3.3 Data collection and analysis

Sixty minutes of footage were reviewed from each camera from each tide, totalling 252 hours of footage filmed across seven months. The 60 minutes began and finished at the same time across all cameras (± 5 minutes as recording was started manually and therefore not simultaneous). The relative abundance of species (MaxN) and the number of species (S) was recorded for each 60 minutes. MaxN is the maximum number of individuals observed in a single frame (Cappo et al. 2004; Cappo et al. 2007; Erickson et al. 2023) over 60 minutes and avoids repeat counts of individuals. The number of species (S) was determined as the total number of species recorded over 60 minutes.

In addition to MaxN and S, the following was also recorded for each organism: total length of observed interaction with rockpool/ seawall, and type and length of behaviour within the interaction (e.g., locomotion, resting, Martinez-Baena et al. 2022, feeding Espadero et al. 2020). Observation began when an organism entered the frame and came within 50 mm of the substrate surface (Campbell et al. 2018; Ushiama et al. 2019) or the seaweed growing on the substrate and ended when the organism left the frame or was out of sight (i.e., behind seaweed) for more than thirty seconds. Organisms that remained in the water column >50 mm away from the substrate surface for the duration of time they were in the field of view and did not make tactical contact with the rockpool/ seawall, were not included in analysis (*sensu* Ng et al. 2021). This was to ensure all behaviours recorded were connected to the rockpool/ seawall.

Types of behaviour included locomotion, resting, feeding and conspecific (Tables 9 and 10). Conspecific behaviour was observed as either reproductive or aggressive in nature. It was not possible to infer exact behaviour other than physical contact for some instances and as conspecific behaviour was relatively infrequent, all conspecific behaviour was pooled into the same category. Feeding for grazing organisms (crabs, prawns) was measured in length of time, whereas feeding for benthic fish was measured as number of bites observed (quantified by contact with mouth to substrate/ seaweed Ushiama et al. 2019; Ng et al. 2021; Taira et al. 2020). Length of behaviour was determined as time spent performing that behaviour for at least 30 seconds.

Table 9. Ethogram of state behaviours (recorded by length of time, one state begins when another has ended if multiple states observed)

Feeding (crabs, prawns)	Organism uses of claws/ chelate legs to gather food items from the benthic environment
Locomotion	Organism swims or crawls around environment
Resting	Organism remains immobile, either on the substrate or in the water column
Conspecific	Organism makes physical contact with a member of the same species. Nested within this are reproductive, predatory and competitive behaviours due to infrequency observed.

Table 10. Ethogram of event behaviours (recorded by number of events, events can occur during state behaviours)

Feeding (fish)	Organism makes brief, rapid contact with mouth to a food item (Morris et al. 2017). This behaviour is recorded regardless of whether the attempt was successful.

To test for statistically significant differences in abundance (MaxN), species richness, and species behaviour between rockpool and seawall habitats over time, linear mixed effect models were run using the "nlme" package (Pinheiro et al. 2020) in R Studio (Version 1.2.1335). For behavioural analyses, only shanny and crabs were used as they were the most frequently observed species in both habitats throughout the study. Assumptions of statistical tests were verified by examination of residuals against fitted model plots, as per Zuur et al. (2009). Where clear lack of normality or heteroskedasticity were identified, transformations were made to the data (log+1 transformations) before further statistical analysis. Habitat (rockpool and seawall) and month (April – October) were fixed factors. To account for repeated measures within the rockpool and seawall, as the same rockpool and seawall sections were surveyed each time, rockpool and seawall 'ID' was included as a random factor. To account for pseudo replication, as two consecutive tidal cycles were surveyed each month, 'replicate' was also included as a random factor, with 'ID' nested within it. Pairwise tests were run using the "emmeans" package (Lenth 2021). To test for similarity between assemblages, a SIMPER analysis was run using Plymouth Routines in Multivariate Ecological Research (Primer-e v.7). To determine any relationship between algae cover and mean interaction time of shanny (*Lipophrys pholis*) and shore crabs (*Carcinus maenas*), two way ANOVAs were run with Algae Percentage Cover and Habitat as factors.

5.4 Results

A total of 8 mobile taxa were observed between April and October 2022 at the study site, including four invertebrates, three cryptobenthic fish and one demersal fish (Table 11). Of these, six were recorded in the rockpools and seven were recorded on the seawall. Three species were only recorded once: the sea slater *Idotea granulosa*, a sea spider (pycnogonida), and the three-spined stickleback fish *Gasterosteus aculeatus*. Two species, European bass *Dicentrarchus labrax* (Figure 18) and the goby *Pomatoschistus* sp., were recorded most months.

Table 11. Mean MaxN per 60 minutes (standard deviation in brackets) of observed species within rockpool and seawall habitats observed at high tide between April and October 2022.

Common name	**Taxonomic name**	**Rockpool**	**Seawall**
Green shore crab	*Carcinus maenas*	0.98 (±0.92)	1.03 (±1.37)
Sea slater	*Idotea granulosa*	0	0.009 (±0.09)
Sea spider	Pycnogonid	0.008 (±0.09)	0
Prawn	*Palaemon* sp.	0.5 (±1.13)	0.4 (±0.56)
Three-spined stickleback	*Gasterosteus aculeatus*	0	0.009 (±0.09)
Shanny	*Lipophrys pholis*	1.26 (±0.97)	1.11 (±0.92)
Goby	*Pomatoschistus* sp.	0.02 (±0.13)	0.23 (±0.46)
European bass	*Dicentrarchus labrax*	0.13 (±0.33)	0.04 (±0.2)
Total		**6**	**7**

Figure 18. A crab (Carcinus maenas) on a control section of seawall (left) and a bass (Dicentrarchus labrax) and crab (C. maenas, partially obscured by seaweed) in a rockpool (right).

5.4.1 Species richness

Within the three artificial rockpool treatments (1, 3, 5 rockpool array), there was very little variation in species richness and no significant difference between treatments (p=0.8501). Pairwise tests revealed a weak significant difference in species richness between 3 and 5 rockpool treatments in June (p=0.0496). As a result, all rockpool treatments have been pooled.

The maximum species richness recorded was 5 on a single occasion in June on the control seawall. No species (0) were recorded on one tidal replicate in April and another in May, and this occurred predominantly in the rockpool habitat in the April tidal replicate and evenly in both habitats in the May tidal replicate. In both habitats, mean species richness remained relatively low throughout the study period, peaking at 3 species in June on the seawall (Figure 19). There was a significant difference between the species richness in habitats over time (p=0.0003). The seawall was significantly more species rich than the rockpools in June, and the rockpools were significantly more species rich than the seawall in September (Table 12, Figure 19).

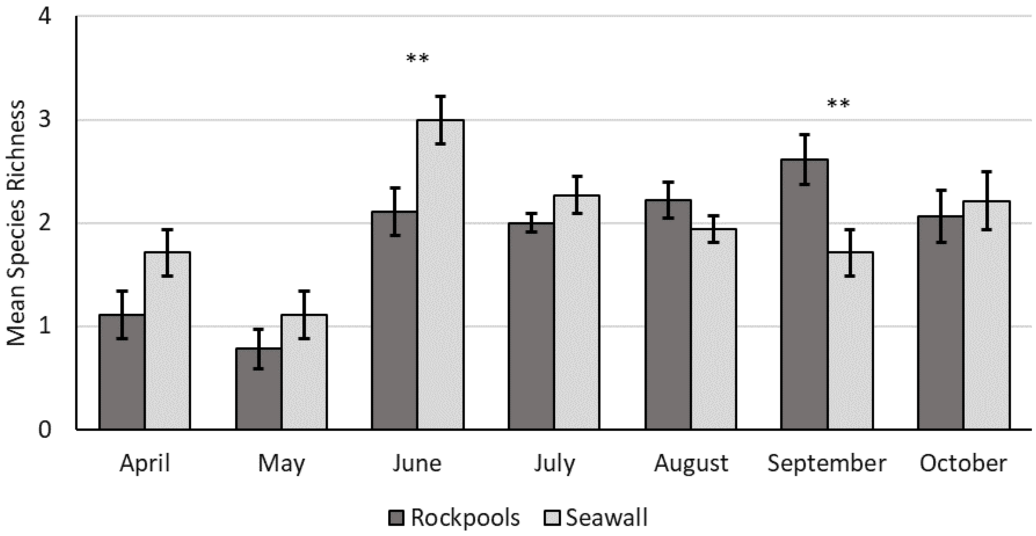


*Figure 19. Mean species richness over time. Statistically significant interactions indicated by * (<0.05), ** (<0.01) and *** (<0.001). Error bars show standard error.*

*Table 12. Pairwise results for species richness (habitat*month). Bold values indicate significant result.*

Factor	Estimate	SE	df	t.ratio	p.value
April	-0.5591	0.304	17	-1.838	0.0836
May	-0.3662	0.304	17	-1.204	0.245
June	-0.9005	0.304	17	-2.962	**0.0087**
July	-0.2908	0.317	17	-0.917	0.3719
August	0.2979	0.308	17	0.967	0.3472
September	0.9423	0.308	17	3.059	**0.0071**
October	-0.1805	0.324	17	-0.557	0.5846

5.4.2 Abundance

There was no significant difference in species abundance between the three rockpool treatments (p=0.4388), though pairwise tests showed a significant difference in MaxN between single and 5 rockpool treatments (p=0.0111) in October. As a result, all rockpool treatments have been pooled.

The maximum abundance recorded was 12 organisms in October, which can be attributed to an aggregation of prawns in the rockpool basin. Mean abundance remained low in the spring months (April, May) before peaking in June for the seawall with 3.9 organisms and peaking in August for the rockpools with 2.9 organisms (Figure 20). There was a significant difference for MaxN between months and habitats and months (Table S17 Supplementary Material). The abundance in both habitat changes over time, with little difference between abundance in habitats in April, May and July, but in June the abundance on the seawall is significantly higher. Conversely, from August to October the reverse is true, with abundance in the rockpools significantly higher (Table 13).

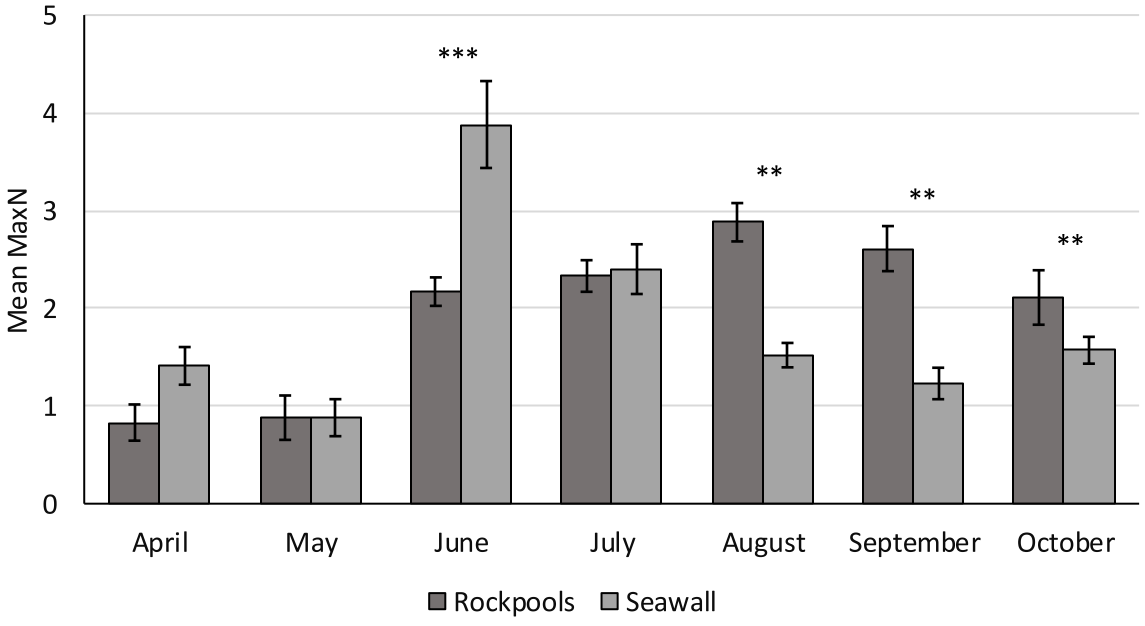


*Figure 20. Mean abundance over time. Statistically significant interactions indicated by * (<0.05), ** (<0.01) and *** (<0.001). Error bars show standard error.*

*Table 13. Pairwise results for MaxN (habitat*month). Bold values indicate significant result.*

Factor	**Estimate**	**SE**	**df**	**t.ratio**	**p.value**
April	-0.57848	0.39	17	-1.483	0.1565
May	0.00902	0.39	17	0.023	0.9818
June	-1.7317	0.39	17	-4.435	**0.0004**
July	-0.05928	0.402	17	-0.147	0.8846
August	1.36115	0.391	17	3.486	**0.0028**
September	1.39223	0.391	17	3.565	**0.0024**
October	1.19755	0.411	17	2.914	**0.0097**

5.4.3 Assemblages

Shanny, prawns and crabs were recorded throughout the study period, and formed the predominant mobile assemblage associated with both the seawall and the rockpools. Gobies first appeared in the seawall habitat in June and were then occasionally observed on the seawall until the end of the study period. One goby was observed on two consecutive tides in September using the same rockpool, but gobies were otherwise absent from the rockpools. Juvenile bass were rarely observed in July in the seawall habitats, before appearing more frequently in both habitats in August, September and October. While their infrequent appearances in the earlier part of the study period were exclusively in seawall habitat, their increased presence in the latter months can be attributed to their visitations of the rockpools.

A SIMPER analysis revealed an average similarity of 43.10% and 45.05% within the seawall and within the rockpools respectively. Between the seawall and rockpools, there was an average dissimilarity of 57.05%, with shannies, crabs and prawns contributing 85.5% to the dissimilarity, and all other species contributing to the remainder. Average similarity between habitats ranged from 58% (September) to 84% (July) over the study period.

5.4.4 Behaviour

At the end of the study period, the mean length of interaction for all organisms at the study site was 101 seconds and ranged between 81 seconds (August) to 194 seconds (October). Apart from June, organisms spent more time interacting with the rockpools than the seawall each month, significantly so between August and October. Only in April and May was there little difference between the rockpools and seawall (Figure 21).

The longest time an organism spent interacting with a habitat was the full hour, and this occurred on four occasions exclusively with shore crabs resting in the rockpools. The longest time an organism spent interacting with the seawall was a crab resting for 2884 seconds, or just over 48 minutes.

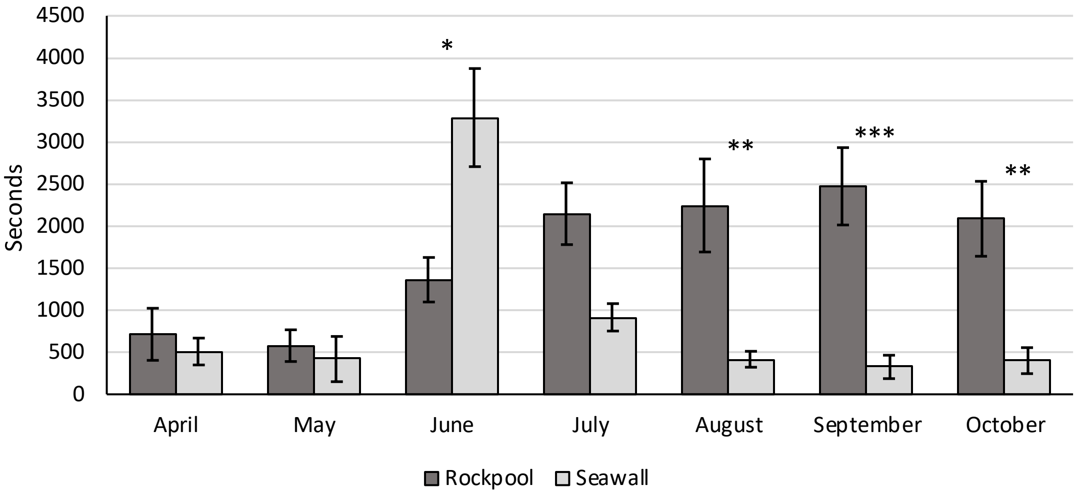


*Figure 21. Mean total interaction times in the rockpools and the seawall across the study period. Statistically significant interactions indicated by * (<0.05), ** (<0.01) and *** (<0.001). Error bars show standard error.*

5.4.4.1 Crabs

There were no significant differences between the three rockpool (1, 3 or 5 rockpools) treatments for all crab behaviours. Pairwise tests found weakly significant differences between single and 3 rockpools in October for crab movement (p=0.0309). As a result, all rockpool treatments have been pooled. There was no relation between algae percent cover and the mean interaction time for crabs (p=>0.05).

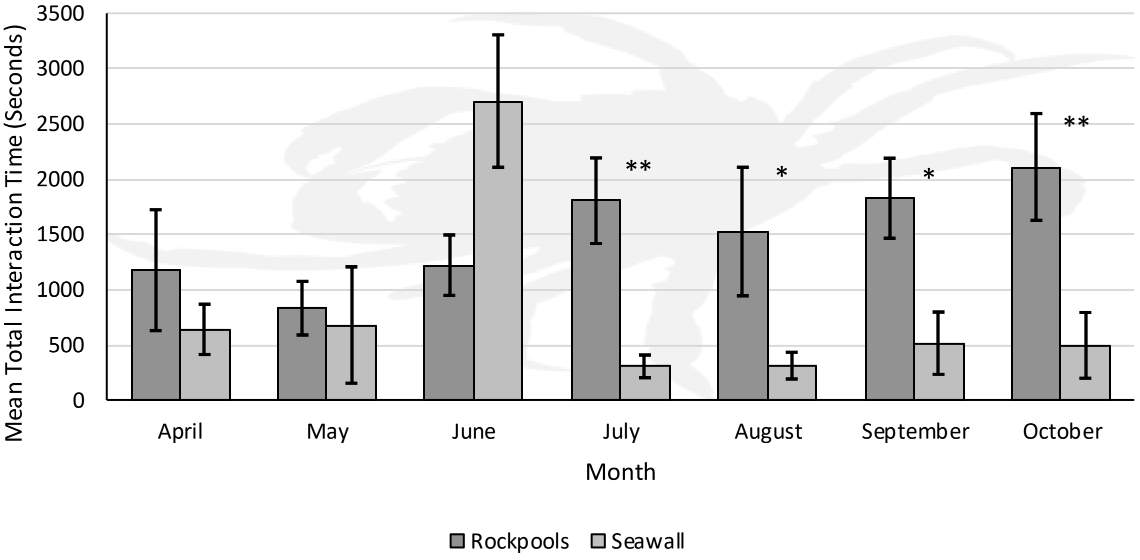


*Figure 22. Mean length of interaction crabs had with both habitats (rockpools, seawall). Statistically significant interactions indicated by * (<0.05), ** (<0.01) and *** (<0.001). Error bars show standard error.*

Total time crabs spent interacting with the habitats was significantly different (Table 9 Appendix) across the study period (habitat*month, p=0.0030). In every month except June, crabs spent on average more time in the rockpools, significantly so from July onwards (Figure 22). Over the study period, crabs spent on average 43% of time resting, 28% of time feeding, 26% of time moving, and 3% of time engaging in conspecific behaviours. Crabs spent over double the amount of time resting in rockpools than they did on the seawall. Conversely, crabs spent almost triple the amount of time engaging in locomotive behaviour on the seawall than in the rockpools. Conspecific behaviour, notably pre-copulatory behaviour, was only observed in the rockpools.

Crabs spent more time feeding in the rockpools than the seawall except for June and October, though this was not significant. There was no significant difference in time spent feeding over the study period (habitat*month, p=0.1080) or between habitats (p=0.9198, Table S17 Supplementary Material). Time spent resting peaked in August with 976 seconds in rockpools and 313 seconds on the seawall in June, and time spent feeding peaked in April for the rockpools 847 seconds and 722 seconds in June for the seawall. Crabs spent significantly more time (habitat, p=0.0006) resting in the rockpools compared to the seawall (Figure 23), including over the study period (habitat*month, p=0.0055, Table S17 Supplementary Material). Time crabs spent moving in both habitats remained below 200 seconds in May, July, August and September, but peaked in the rockpools in October with 299 seconds and on the seawall in June with 1084 seconds. There was no significant difference in time spent moving (habitat, p=0.0531) between habitats (Figure 23), including over the study period (habitat*month, p=0.2325, Table S17 Supplementary Material), though crabs spent significantly more time moving on the seawall than the rockpools in June (Table 14). June was particularly notable, as large numbers of small crabs were observed on the seawall and is reflected in overall MaxN trends (Figure 20).

Table 14. Pairwise results for crab behaviour in artificial rockpool versus seawall for each month. Bold values indicate significant result.

Factor	Estimate	SE	df	t.ratio	p.value
Total Time					
April	0.2617	0.794	17	0.329	0.7458
May	1.6483	0.86	17	1.916	0.0723
June	-0.6981	0.55	17	-1.269	0.2216
July	1.7983	0.571	17	3.148	**0.0059**
August	1.3841	0.644	17	2.149	**0.0463**
September	1.999	0.763	17	2.621	**0.0179**
October	2.2595	0.72	17	3.138	**0.006**
Moving					
April	-1.03001	1.121	17	-0.919	0.3709
May	0.90739	1.216	17	0.746	0.4656
June	-1.98587	0.766	17	-2.593	**0.0189**
July	-0.80305	0.796	17	-1.009	0.327
August	0.53638	0.902	17	0.595	0.5597
September	-1.3505	1.075	17	-1.257	0.2259
October	0.02023	1.013	17	0.02	0.9843
Resting					
April	-1.8627	1.58	17	-1.179	0.2548
May	2.5707	1.73	17	1.486	0.1557
June	-0.1682	1.07	17	-0.158	0.8764
July	3.8463	1.11	17	3.472	**0.0029**
August	2.4374	1.27	17	1.925	0.0712
September	4.3246	1.52	17	2.841	**0.0113**
October	4.7558	1.43	17	3.332	**0.0039**

Feeding					
April	1.9113	1.47	17	1.3	0.211
May	2.6792	1.602	17	1.672	0.1128
June	-1.5002	0.995	17	-1.508	0.1498
July	1.505	1.034	17	1.455	0.1638
August	0.9449	1.178	17	0.802	0.4336
September	0.224	1.412	17	0.159	0.8758
October	-1.5747	1.327	17	-1.186	0.2518

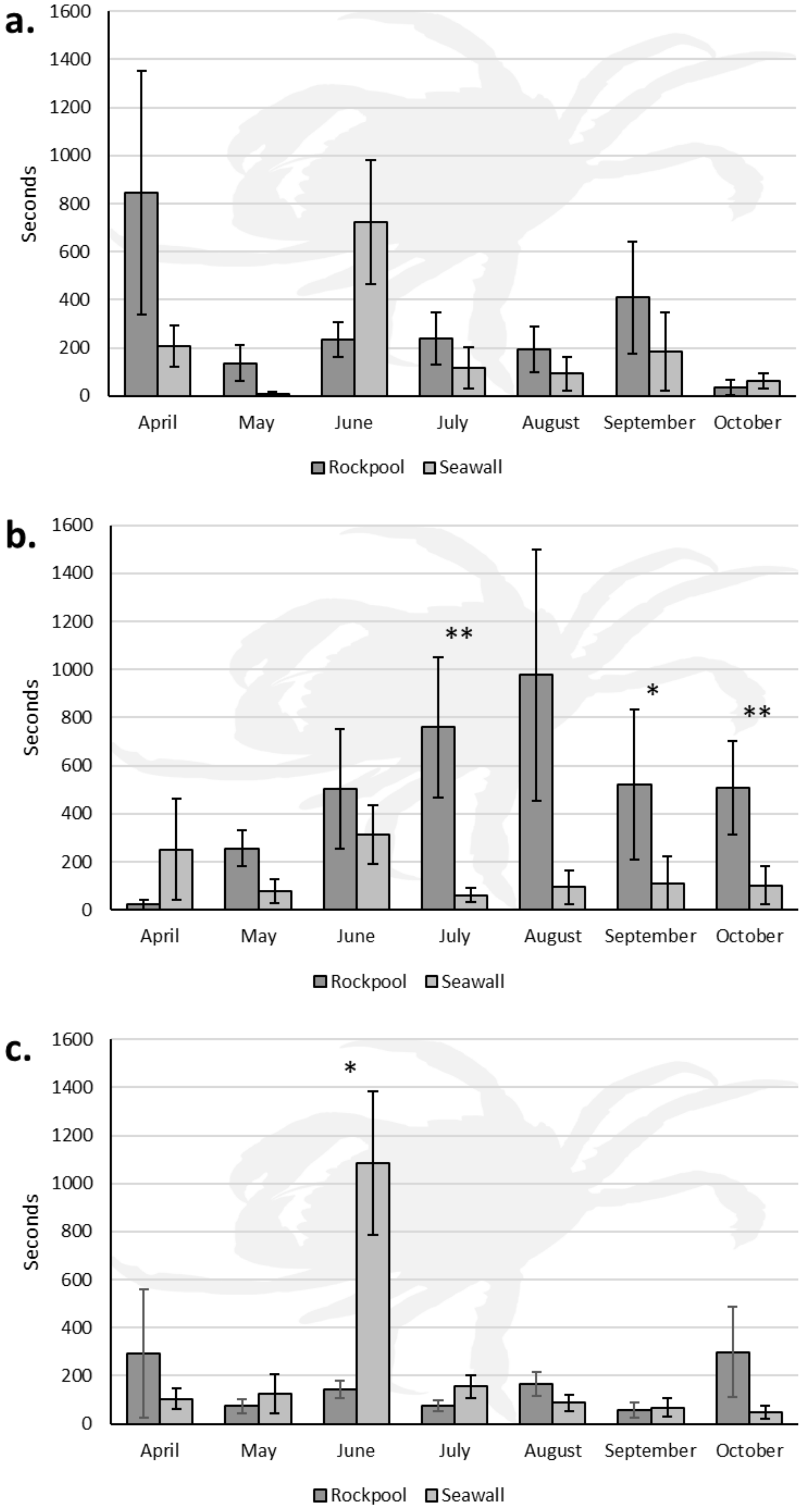


*Figure 23. Mean length of time crabs spent feeding (a), resting (b), and moving (c) in both habitats (rockpools, seawall). Statistically significant interactions indicated by * (<0.05), ** (<0.01) and *** (<0.001). Error bars show standard error.*

5.4.4.2 Shanny

There were no significant differences between the three rockpool (1, 3 or 5 rockpools) treatments for all shanny behaviours, except for treatment*month for moving behaviour. Analyses for shanny behaviour omitted data from June as shanny did not interact with 3 rockpool treatments during this month. For shanny movement between single and 3 rockpools in April (p=0.0449) and between 3 and 5 rockpools in September (p=0.0137), there were weak significant differences. Total interaction time for shanny was also significantly different between 3 and 5 rockpools in October (p=0.0088). As a result, all rockpool treatments have been pooled. There was a weakly significant relationship between algae percent cover and mean interaction time for shanny (p=0.012) compared to a moderately significant relationship between habitat and mean interaction time (p=0.0043).

Total time shanny spent interacting with the habitats was significantly different (habitat, p=0.0042) (Figure 24), including across the study period (habitat*month, p=<0.0001, Table 9 Appendix). Except for April, August and September, where mean shanny interaction time was significantly greater in the rockpools than the seawall (Table 15), there is little difference in shanny interaction time between the habitats (Figure 24).

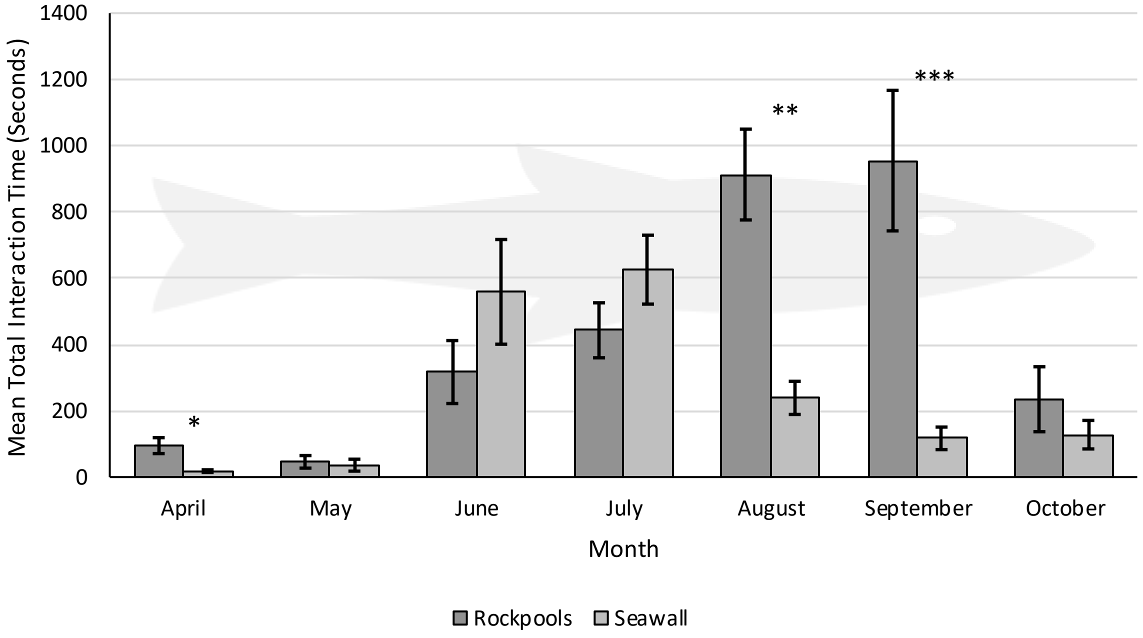


*Figure 24. Mean total length of interaction shanny had with both habitats (rockpools, seawall). Statistically significant interactions indicated by * (<0.05), ** (<0.01) and *** (<0.001). Error bars show standard error.*

Over the study period, shanny spent on average 73% of time moving, 26% of time resting, and only 2% of time engaging in conspecific behaviours. Shanny spent 72% and 75% of their time moving on the rockpools and the seawall respectively. Similarly, there is little difference in the time spent resting in both habitats, with 27% and 24% of resting time spent on the rockpools and seawall respectively. Conspecific behaviour was only observed in the rockpools.

Shanny spent significantly more time resting in the rockpools than the seawall in August (Figure 25, Table 15). Time spent resting peaked in September with 521 seconds in rockpools and 228 seconds on the seawall in June. Time shanny spent resting within the habitats was not significantly different (habitat, p=0.1508, Table 9 Appendix), but time spent resting was weakly significant when factoring in month (habitat*month, p=0.0339).

Time spent engaging in locomotive behaviour in both habitats increased in the summer months but declined to pre-summer levels in September and October. Locomotive behaviour peaked in the rockpools in August with 629 seconds and on the seawall in June with 228 seconds. Time shanny spent moving within the habitats was significantly different (habitat, p=0.0086, Table 9 Appendix), including over the study period (habitat*month, p=<0.001), with significantly more time spent moving in the rockpools in August and October than the seawall (Figure 25, Table 15).

Table 15. Pairwise results for shanny behaviour in artificial rockpool versus seawall for each month. Bold values indicate significant result.

Factor	Estimate	SE	df	t.ratio	p.value
Total Time					
April	1.4712	0.622	17	2.365	**0.0302**
May	0.3504	0.685	17	0.512	0.6155
June	-0.792	0.486	17	-1.628	0.1219
July	-0.286	0.44	17	-0.65	0.5245
August	1.5373	0.427	17	3.596	**0.0022**
September	2.2539	0.48	17	4.691	**0.0002**
October	0.1966	0.477	17	0.412	0.6852
Moving					
April	0.8006	0.652	17	1.229	0.236
May	0.0478	0.716	17	0.067	0.9476
June	-0.8584	0.514	17	-1.671	0.1131
July	-0.4353	0.466	17	-0.934	0.3633
August	1.6732	0.453	17	3.692	**0.0018**
September	2.5729	0.507	17	5.071	**0.0001**
October	0.4916	0.504	17	0.976	0.3428
Resting					
April	0.583	1.205	17	0.484	0.6346
May	0.0402	1.326	17	0.03	0.9762
June	-0.7726	0.945	17	-0.818	0.4247
July	0.6162	0.855	17	0.721	0.481
August	3.0775	0.831	17	3.704	**0.0018**
September	0.9101	0.933	17	0.976	0.3429
October	-0.4944	0.926	17	-0.534	0.6002

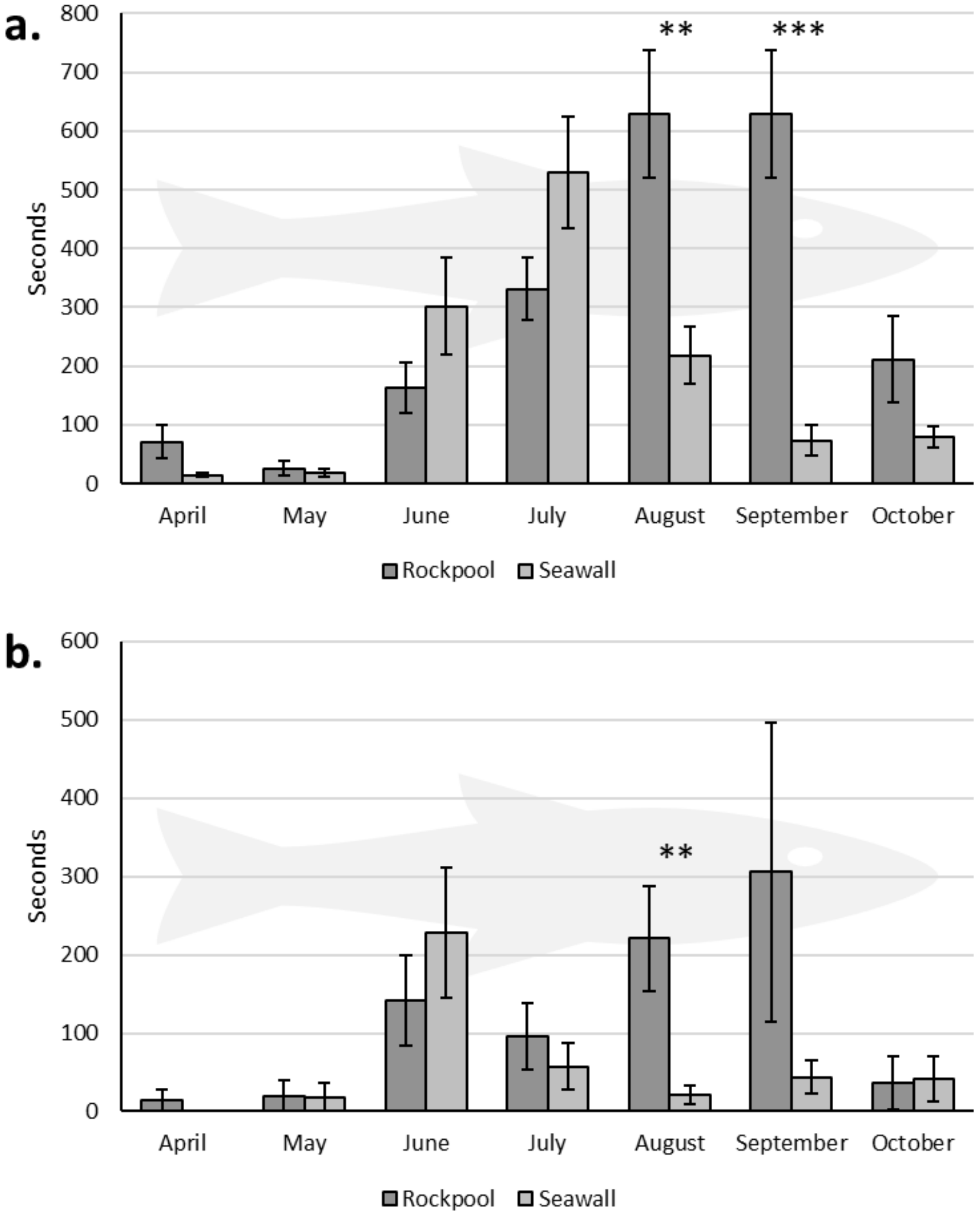


*Figure 25. Mean length of time shanny spent moving (a) and resting (b) in both habitats (rockpools, seawall). Statistically significant interactions indicated by * (<0.05), ** (<0.01) and *** (<0.001). Error bars show standard error.*

The mean number of bites observed peaked in September, and throughout the study period more bites were observed on the seawall than the rockpool, with no bites observed on the rockpools during April and May. Shanny bites differed significantly between habitats (p=0.0425) and between rockpools and seawall over time (habitat*month p=<0.0001, Table SS17 Supplementary Material). Significantly more bites were observed in June, July and September on the seawall than on the rockpools (Figure 26, Table 16). Shanny were observed to be feeding predominantly on barnacles, though on a handful of occasions were observed to pluck small littorinid snails from the substrate before dropping them. On one occasion, a shanny preyed on a small crab in the rockpool.

Table 16. Pairwise results for total number of shanny bites in artificial rockpool versus seawall for each month. Bold values indicate significant result.

Bites	Estimate	SE	df	t.ratio	p.value
April	-0.1244	0.366	17	-0.34	0.7382
May	-0.3161	0.403	17	-0.784	0.4439
June	-0.9742	0.286	17	-3.404	**0.0034**
July	-1.0584	0.259	17	-4.087	**0.0008**
August	0.0532	0.251	17	0.211	0.8351
September	0.7236	0.283	17	2.559	**0.0203**
October	-0.4713	0.28	17	-1.68	0.1112

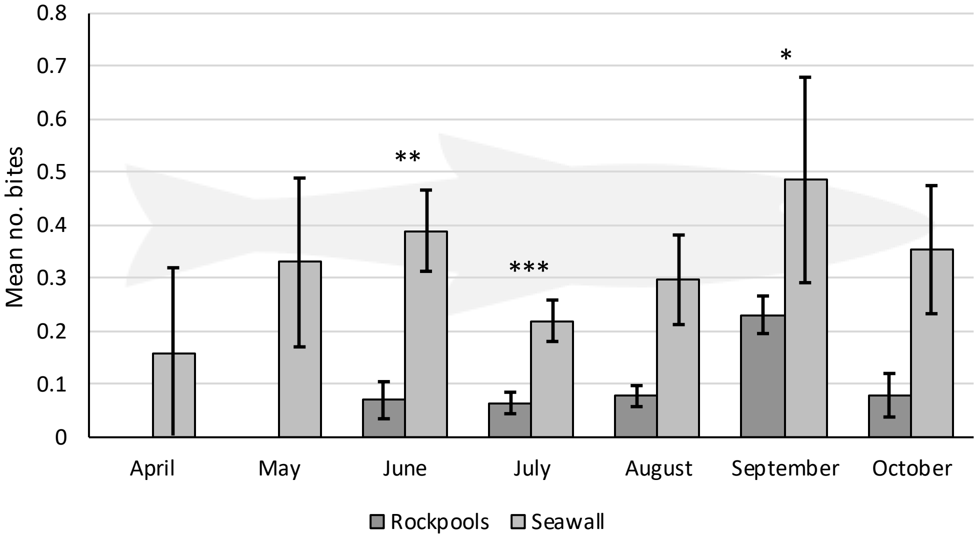


*Figure 26. The mean number of bites performed by shanny fish in a 60 minute period in both habitats over time. Error bars show standard error. Statistically significant differences indicated by * (<0.05), ** (<0.01) and *** (<0.001)*

5.5 Discussion

The abundance and species richness of mobile fauna utilising both the rockpools and seawall at high tide is not significantly different overall, which is consistent with the existing literature (Mercader et al. 2017; Morris et al. 2017). Yet, organisms generally spent more time in the rockpools than on the seawall, but this was not consistent over the study period, or consistent within crabs or shannies.

Crabs spent more time throughout the study period in the rockpools than the seawall, particularly to rest. Although determining size of organisms was beyond the scope of this study, crabs that used the rockpools were generally of a larger size than the crabs that predominantly used the seawall. This was particularly evident in June, when a relatively high abundance of small crabs (< 2cm) was recorded on the seawall, both feeding and moving over the extensive coverage of barnacles. Juvenile crabs in lab experiments have shown to preferentially feed on barnacle cirri over littorinid snails (Rangeley and Thomas 1987) and the partial predation on barnacles in this study is consistent with that observed by Burrows et al. (1999). Although barnacles have been recorded inside the rockpools, their coverage is comparatively minimal. Consequently, the rockpools may not have provided sufficient food for the high abundance of juvenile crabs compared to the barnacle-covered seawall during June. In addition, the high percentage cover of canopy-forming fucoid algae on the rockpools may have obscured organisms from view, leading to an underestimation of smaller crabs in the rockpools. Juvenile crab carapaces often feature white pigmentation that aids camouflage on a light-coloured, barnacle-dominated substrate (Todd et al. 2006; Stevens et al. 2014). This pigmentation usually disappears in older, larger crabs (Bedini 2002; Todd et al. 2006) that prefer dark, macroalgal-dominated habitats (Hogarth 1978; Orlosk et al. 2011; Barr and Elwood 2011; Twort and Stevens 2023) which has been shown to provide spatial refuge from predation by gulls (Dumas and Witman 1993). The dimly lit rockpool basins shadowed by the overlying algae canopy would have provided such spatial refuge and supports the observations in this study of smaller, lighter crabs generally being observed on the barnacle-dominated seawall and the larger, darker crabs on the macroalgae-dominated rockpools. Crabs spent significantly more time resting in the rockpools than the seawall. This was usually in the basin, below the relief of the rockpool rim, or occasionally on the rim but lodged in the interstices between fucoid holdfasts. This suggests that the rockpools, through enhanced epilithic complexity, provide sufficient shelter that is otherwise not present on the seawall. Crothers (1968) determined that crabs are unlikely to be found resting in open and homogenous habitat, such as the seawall, and are more likely to be sheltering in crevices. Rockpools were the only habitat where reproductive behaviour was observed, with male crabs gripping females in a pre-copulatory position in the rockpool basin. The mobility of male crabs in pre-copulation is limited (Adams and Greenwood 1985; van der Meeren 1994) and so the more sheltered nature of rockpools may permit pre-copulatory pairs to minimise movement that would otherwise be necessary to prevent dislodgement on a turbulent and exposed seawall. As retrofitted rockpools can disrupt hydrodynamics on a vertical seawall (Salauddin et al. 2021), this may benefit crab foraging efficiency where flow rates are reduced (Robinson et al. 2011). As observed by Sheehan et al. (2010) with crab-tiles, artificial crab habitats on intertidal soft sediment, crabs may spend more time in rockpool habitats at high tide to guard it for their use as refugia at low tide.

Shanny spent significantly more time in the rockpools for three of the months. Goncalves and Almada (1998) found that shanny minimized the time spent outside of their nest territories, which reduces the time spent in turbulent intertidal conditions, such as on the seawall, where the risk of physical injury and dislodgement is high. Conversely, shanny fed significantly more on the seawall than the rockpools. It is clear in this study that the cryptobenthic shanny use the rockpools as shelter but prefer the feeding opportunities of the seawall. This is consistent with the results of Ushiama et al. (2019) which found that feeding activity of cryptobenthic fish was greater on flat tiles than structurally complex tiles. The prey available on the seawall better supports the preferred shanny diet (Maze et al. 1999), particularly for the relatively small shannies observed during this study (Faria and Almada 2008). Shanny are visual feeders (Gibson 1972; Davenport et al. 2023), and so feeding on the well-illuminated seawall may explain their preference. Fish predation pressure on sessile invertebrates increases in areas of lower structural complexity (Bolton et al. 2018). There was a weakly significant relationship between algae cover and mean interaction time with shanny, but as high algae cover was strongly associated with the rockpools, it is not possible to disentangle which factor is most responsible, though it is likely a combination. Shanny were observed navigating the relatively flat and featureless surface of the seawall by resting in the interstices of barnacle tests where the concrete substrate was exposed or on the leeward side of limpets, as the majority of those observed were relatively small.

Shanny will navigate to dark, shadowy crevices when threatened (Dodd et al. 2000). Indeed, the greater algal canopy cover and structural complexity of the rockpools offers cryptobenthic fish enhanced predator avoidance (Schofield 2003; Gregor and Anderson 2016), and the crevices and interstices that occur on the rockpool substrate provide habitat more suited for shanny body size (Randall 1963; Hixon and Beets 1989; Nunes et al. 2019). It was observed in this study, though not formally recorded, that shanny would squeeze into the narrow gap between the back of the rockpool and the seawall. It has been suggested that shanny activity is greatest at flooding or high tide in the morning, when shanny are likely to be hungriest as they do not feed at night (Burrows et al. 1999; Faria and Almada 2006). Therefore, later diurnal high tides may bring about differences in the drivers of shanny activity and consequently their interactions with the rockpools and the seawall. Shanny show fidelity to a network of shelter sites and will access them based on a hierarchy of dominance mediated by body size (Almada et al. 1983; Faria and Almada 2001). The majority of shanny observed in this study were relatively small, with very few incidences of agonistic behaviour which suggests that although the rockpools may increase the likelihood of intraspecific interaction through provision and concentration of shelter spaces, shelter spaces are not so few that competition is fierce.

The recording of these behaviours in the rockpool demonstrate the importance of habitat provision on habitat-poor artificial coastal structures, particularly when it may boost mating opportunities and reduce predation risk. However, it is evident that the seawall also provides habitat resources, such as food provision for juvenile crabs and shanny, which highlights the importance of habitat heterogeneity and patchiness of habitat features and complexity (Paxton et al. 2017; Ushiama et al. 2019; Hall et al. 2021; Bishop et al. 2022) to support species' different requirements throughout their life history. Canopy-forming fucoid growth was strongly associated with the rockpool habitat, while high coverage of barnacles was strongly associated with the seawall. The high abundance of fucoid algae on the rockpools was likely by virtue of the rugose texture and horizontal orientation on the rockpool rim, where fucoid holdfasts were predominantly fixed, providing a sufficient settlement surface (Fletcher and Callow 1992) and adequate sun exposure. This topside algal colonisation has been observed with artificial rockpools of a similar commercial model elsewhere in the UK (Drakard et al. 2023) and artificial reefs (Herbert et al. 2017). Vegetated/ complex and non-vegetated/ less complex habitats are important for shore crabs at different life history stages (Amaral et al. 2009) and so the vegetated complexity of the rockpools mixed with the comparatively structurally simple seawall may support shore crabs from juveniles to adults.

Further study should focus on the connectivity between ecological enhancement habitats and the substrate they are on. This study was limited by the camera orientation that only permitted recording of the topside of the rockpools, when it was evident that mobile fauna was moving back and forth over the rim and potentially using the underside of the rockpools and gaps between closely grouped rockpools. The set-up also potentially excluded interactions from demersal fish which may have been attracted to the rockpools or interacted with the rockpool underside but did not appear on camera. Replication at a larger scale, for example filming away from but looking back at the rockpool and the surrounding seawall, would provide a greater understanding as to the wider spatial impact of the rockpools. We did not pursue this due to logistical challenges and the very public nature of the study site. It is clear from existing literature that shanny and shore crab behaviour may differ between day and night (Davenport et al. 2023). Therefore, elucidating habitat use of the seawall and rockpools at night would be beneficial as to date no behavioural studies of ecological enhancements have been performed at night.

Lack of significant differences between the rockpool treatments (1, 3 or 5 rockpools) may have been due to the camera recording only a single, central rockpool, whereas there may have been greater activity at the edges of the array. Yet it may also indicate that these multiple-pool arrays do not offer any additional habitat benefits other than replicating that of individual pools. Therefore, the array, as designed, may not create habitat complexity above that of the sum of the individual component pools.

Intertidal ecological enhancement often focuses on shelter provision at low tide and incorporates features that are skewed to that requirement. However, some of the features seen in subtidal artificial reefs (deep holes, tunnels and other 'swim-through' features) have relevance in the intertidal at high tide, despite their potentially lower habitat value at low tide. It was noted by Faria and Almada (2001) that crevices and holes for larger adult shanny (15-20 cm) were usually absent from natural rocky shores, and so there may be value in providing larger holes to support individual taxa throughout their life histories. As demersal fish also use artificial coastal structures, as observed in this study and others (Herbert et al. 2017; Hall et al. 2021), there is justification for enhancement features that cater to their habitat needs (see Morris et al. 2018). Providing enhancement that encourages epilithic complexity, such as canopy-forming macroalgae, will provide additional habitat value that the enhancement alone cannot. For cryptobenthic fish, holes and cracks that match their body sizes would benefit their survival. With artificial rockpools, this could be achieved by varying the space between the back of the rockpools and the seawall, as has been incorporated in Sydney by Bishop et al. (2022) and *Artecology Ltd.* (personal communication) creating different habitat spaces. However, it is important any ecological enhancement is implemented with the advice and recommendation of a local ecologist, particularly on urbanised coasts. Enhancement should be considered on a site-by-site basis, as features are not one-size-fits-all, and enhancement should not be used to greenwash construction schemes (Firth et al. 2020).

5.6 Conclusion

Our study has shown that while intertidal ecological enhancements can sometimes have limited impact on the abundance and species richness of mobile fauna at high tide, they provide crucial habitat that allow organisms, such as crabs and cryptobenthic fish, to rest, mate and find refuge. It is evident that both flat seawalls predominantly covered in barnacles and artificial rockpools predominantly covered in macroalgae are both important for the survival of intertidal mobile fauna. A patchwork of different surfaces and features would therefore best support these species at high tide, in addition to adding features more commonly seen on subtidal artificial reefs, such as deep crevices, holes and tunnels.

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5.8 Supplementary material

Table S17. Main tests for species richness, MaxN, crab behaviour, and shanny behaviour between rockpools and seawall. Bold values indicate significant result.

Factor	numDF	denDF	F-value	p-value
Species Richness				
Habitat	1	17	0.8841	0.3603
Month	6	193	14.9665	**<.0001**
Habitat*Month	6	193	4.4429	**0.0003**
MaxN				
Habitat	1	17	1.5482	0.2303
Month	6	190	15.3463	**<.0001**
Habitat*Month	6	190	9.6371	**<.0001**
Crab - total time				
Habitat	1	17	8.2944	**0.0104**
Month	6	100	2.68	**0.0187**
Habitat*Month	6	100	3.5793	**0.003**
Crab - moving				
Habitat	1	17	4.3229	0.0531
Month	6	100	5.858	**<.0001**
Habitat*Month	6	100	1.3738	0.2325
Crab - resting				
Habitat	1	17	17.53308	**0.0006**
Month	6	100	0.89002	0.5053
Habitat*Month	6	100	3.27588	**0.0055**
Crab - feeding				
Habitat	1	17	0.01045	0.9198
Month	6	100	4.62324	**0.0003**
Habitat*Month	6	100	1.79356	0.108
Shanny - total time				
Habitat	1	17	10.9271	**0.0042**
Month	6	132	17.7866	**<.0001**
Habitat*Month	6	132	5.7804	**<.0001**
Shanny - moving				
Habitat	1	17	8.8074	**0.0086**
Month	6	132	18.1956	**<.0001**
Habitat*Month	6	132	6.5308	**<.0001**
Shanny - resting				
Habitat	1	17	2.26389	0.1508
Month	6	132	5.07633	**0.0001**
Habitat*Month	6	132	2.35761	**0.0339**
Shanny - bites				
Habitat	1	17	4.81061	**0.0425**
Month	6	132	5.91285	**<0.0001**
Habitat*Month	6	132	5.42803	**<0.0001**

Chapter 6 – Vertical arrays of artificial rockpools on a seawall provide refugia across tidal levels for intertidal species in the UK.

This chapter is currently being reviewed for publication.

6.1 Abstract

Eco-engineering of coastal infrastructure aims to address the insufficient intertidal habitat provided by coastal development and flood defence. There are numerous ways to enhance coastal infrastructure with habitat features, but a common method involves retrofitting artificial rockpools. Often these are 'bolt-on' units that are fixed to existing coastal infrastructure but there is a paucity of literature on how to optimise their arrangement for biodiversity.

In this study, 24 artificial rockpools were installed at three levels between High Water Neaps and Mean Tide Level on a vertical concrete seawall on the south coast of the UK. The species abundance of the rockpools and adjacent seawall were surveyed at low tide for 2 years following rockpool installation and compared. Over the course of the study, sediment had begun to accumulate in some of the rockpools. At the 2-year mark, the sediment was removed and assessed for macrofauna. Algal biomass of the seawall and rockpools was estimated using previously obtained dry weight values for the dominant algae taxa.

After 2 years, it was determined that artificial rockpools successfully increase species richness of seawalls, particularly at higher tidal levels where water-retaining refugia are crucial for many species. The rockpools hosted 37 sessile taxa and 9 sessile taxa were recorded on the seawall. Rockpools increased the vertical elevation for brown canopy-forming seaweeds by providing better attachment surfaces. Although the retained sediment only hosted 3 infaunal species, it was observed to provide shelter for shore crabs during surveys. As sea levels and ocean and air temperatures continue to rise, vertical eco-engineering arrangements will play a crucial role in allowing species to migrate up the tidal zone, negating habitat loss and localised extinction.

6.2 Introduction

Ecological enhancement (also referred to as eco-engineering) is used to add habitat to artificial coastal and marine structures (Bergen et al. 2001; Mitsch and Jorgensen 2003; Odum and Odum 2003) which are ecologically poor surrogates for natural hard substrate environments (Connell and Glasby 1999; Chapman 2003; Moschella et al. 2005; McKinney 2006; Glasby et al. 2007; Vaselli et al. 2008; Pister 2009; Firth et al. 2013; Earp et al. 2023). There are numerous ways this can be achieved which are well summarised (Naylor et al. 2017; Strain et al. 2017; O'Shaughnessy et al. 2020; Evans et al. 2021) but examples include drill-cored or hammered pits in rip-rap rock armour (Firth et al. 2014; Evans et al. 2016; Ostale-Valriberas et al. 2018; Chee et al. 2020), standalone pre-cast intertidal reef blocks and pools (Firth et al. 2014; Perkol-Finkel and Sella 2015; Waltham and Sheaves 2018), habitat features integrated or recessed within the structure through bricks or form liners (Chapman and Blockley 2009; Chapman and Underwood 2011; Firth et al. 2014), and retrofitted tiles, rockpools and panels (Browne and Chapman 2011; Browne and Chapman 2014; Morris et al. 2017; Hall et al. 2019; MacArthur et al.2019; Bishop et al. 2022; Kosova et al. 2023).

Retrofitted (installed after the construction) 'bolt-on' artificial rockpools are commonly used to add ecological enhancement to existing artificial coastal structures (Naylor et al. 2017), particularly for research purposes (Browne and Chapman 2011; Morris et al. 2017; Morris et al. 2018; Hall et al. 2019;

Drakard et al. 2023). These precast, 'off-the-shelf' units are now available on a commercial basis (Evans et al. 2019). However, there is a paucity of literature on how the arrangement (i.e., number, pattern/ configuration) of retrofitted interventions can impact species abundance outcomes (but see Loke et al. 2019).

Optimising the arrangement of intertidal interventions is crucial for incorporating ecological enhancement at scale, as it informs engineers where interventions should be focussed and where they may be less necessary. Tidal height of rockpools determines how long they remain emersed at low tide, with those higher in elevation remaining emersed for longer (Martins et al. 2007; Firth et al. 2013). Consequently, the physico-chemical parameters of the retained water are impacted by exogenous abiotic factors (solar radiation, evaporation, pool area/ depth/ volume) (Daniel and Boydon 1975; Truchot and Duhmanel-Jouve 1980; Huggett and Griffiths 1986; Metaxas and Scheibling 1993; White et al. 2014), biological processes (photosynthesis, respiration), and species interactions (Underwood and Jernakoff 1984; Benedetti-Cecchi et al. 2000). Rockpools experience greater variations of these physico-chemical parameters than the sea (Morris and Taylor 1983) but a lower magnitude than that of the emergent substrate (Metaxas and Scheibling 1993). A rockpool at a higher tidal elevation is likely to experience more extreme temperatures, pH, and salinity than rockpools at a lower tidal elevation that are emersed for less time (Little et al. 2009; Legrand et al. 2018). As a result, the biota they support is generally limited to species that are especially tolerant of more hostile conditions, such as *Ulva* sp. green algae (Legrand et al. 2018; Hall et al. 2019). Tidal height is an important factor in deciding where interventions should be placed within an artificial coastal structure or a flood defence scheme (Firth et al. 2016; Naylor et al. 2017). Therefore, maximising biota on a vertical seawall may be more effective at moderate tidal elevations (Firth et al. 2013). Installing retrofitted interventions may alter the distribution of species on the existing structure, for example through the provision of microhabitats favoured by grazers (Fairweather 1988; Aguilera et al. 2014; Aguilera et al. 2022) or by ameliorating temperature and desiccation stressors by creating shaded, damp areas (Meager et al. 2011; Baxter et al. 2023). Some studies have been conducted on artificial rockpools along a vertical gradient (Browne and Chapman 2014) in the northern hemisphere (Hall et al. 2019; Bone et al. 2022) but independence between levels and replication of rockpools was low. Otherwise, studies have favoured a horizontal array (Browne and Chapman 2011; Morris et al. 2017; Morris et al. 2018; Drakard et al. 2023). In this context, an experiment was undertaken to determine the impact of vertical distribution and tidal height on the colonisation of 'bolt on' rockpools on a vertical seawall. As part of the Marineff Project 24 artificial rockpools (Vertipools) were installed in Hamble Harbour. We aimed to determine how tidal height impacts species abundance on the seawall and artificial rockpools and hypothesised that:

Species abundance will be greater on the artificial rockpools compared to the adjacent seawall.

6.3 Method and materials

The site was based in Hamble Harbour on the River Hamble (50.810179, -0.5492762), Southampton Water on the south coast of England (Figure 27), a highly modified, muddy estuary very popular for sailing with over 3000 moorings. It is sheltered and protected from prevailing south-westerly winds by the Isle of Wight and its location within Southampton Water. Nearby intertidal habitats are primarily soft-sediments and those provided by artificial structures, such as seawalls. The rockpools were affixed to a >100-year-old intertidal vertical concrete seawall which in some areas, due to its age, had a slight positive camber. The seawall is western-facing and can be accessed at low tide on foot (Figure 28).

Visual assessment of the seawall biotic community prior to rockpool installation indicated it was species poor, with some old brown algae *Ascophyllum nodosum* growth dominating the bottom quarter of the wall. At the seawall toe was thick mud and occasional rubble which provided attachment substrate for fucoids and *A. nodosum*. Permission to install the rockpools was provided by the asset owners Warsash Sailing Club and Hamble Harbour Authority.

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Figure 27. The rockpool location in Hamble Harbour on the south coast of the UK.

Figure 28. The seawall in 2017 prior to rockpool installation.

The artificial rockpools were handmade by *Artecology Ltd*. using Vicat Prompt cement, sharp sand ballast (≤ 10 mm) and potable water. They were pre-cast in standardised moulds to the dimensions shown in Figure 29. A short video of the rockpool manufacture may be viewed here (youtu.be/mX0YXjWotWE?si=ubOEqQcCeMfpMh6p). The rim and rockpool interior were made with a stippled texture, and the exterior was finished with concave hemispherical pockets achieved using a bubble wrap form liner.

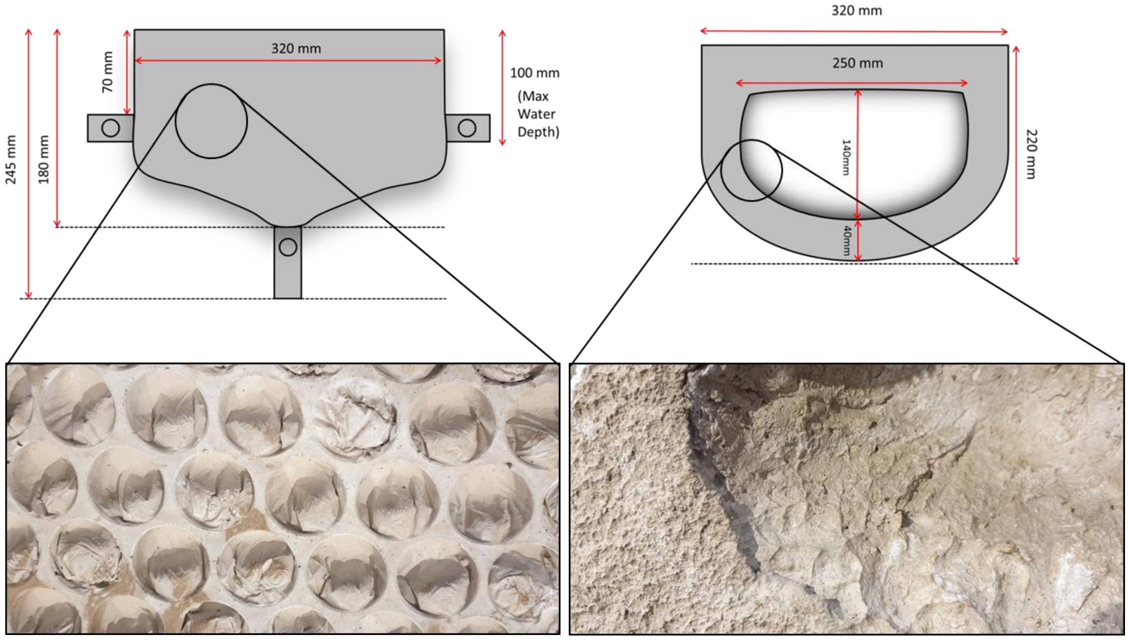


Figure 29. The artificial rockpool dimensions.

Prior to installation the seawall substrate was scrubbed clear of fouling organisms to allow the rockpools to sit flush against the seawall surface. A stainless-steel bracket and hardware were used to fix the rockpools to the seawall in eight columns of three, with the bottom rockpool at mean tide level, the top rockpool at high water neap tide level, and the middle rockpool installed equidistant between both top and bottom rockpools. A short video of the rockpool installation method may be viewed here (youtu.be/QEuiww85F2Y?si=RNdOhDXdsuP4W0SK). Each column was installed within a section of seawall separated by equally spaced buttresses to maintain independence. Installation was completed in October 2020.

Monitoring surveys were undertaken in the early to mid-afternoon on an ebbing tide at 1-, 6-, 9-, 12-, 15-, 18-, 21- and 24-month intervals following completion of installation. During surveys, species abundance data was obtained for each rockpool through visual assessment of percentage cover and numeric counts of organisms to species level wherever possible, with the rockpool interior (pool of water) and exterior (rockpool underside) surveyed separately (Table 18). High resolution photos were taken using a Sony A7R3 CDC camera (Sony FE 24-70mm zoom lens) permitting high quality data collection of photo-quadrats. For each rockpool a photo-quadrat of the seawall was taken either side of the rockpool at the same tidal level (Table 18). During surveys, the seawall was characterised with high abundances of *Ascophyllum nodosum* at lower tidal levels and *Ulva* spp., *Blidingia minima* at higher tidal levels, but otherwise consisted of bare concrete and no understorey algal turf was observed. Due to a high abundance of canopy-forming algae obscuring the underlying seawall at the lower tidal level, it was not possible to ascertain the presence of mobile fauna from seawall photo-quadrats and therefore mobile fauna, such as snails and limpets, have been excluded from data analysis. All results use data derived from sessile species abundance only. A list of mobile fauna species identified in the artificial rockpools was collated separately.

Table 18. The quadrat size and number of quadrat replicates recorded in each habitat per rockpool.

Habitat	**Quadrat Size**	**Replicates**
Artificial Rockpool - Interior	25 cm^2	**1** – the whole pool
Artificial Rockpool - Exterior	25 cm^2	**2** – one on each exterior half
Seawall	25 cm^2	**2** – one either side of the rockpool

After it became evident that sediment accretion was occurring in some of the rockpools, extra care was taken not to disrupt the sediment during surveys to avoid disturbance to infauna. At the final 24-month interval survey all sediment was removed from each rockpool (n=24) for macrofaunal analysis and stored in formalin in labelled watertight zip-lock bags for analysis. Sediment samples were later rinsed of formalin under a fume hood and macrofauna were picked under a Leica stereomicroscope and identified to highest taxonomic resolution. To determine how much biomass grows on the rockpools compared to the seawall, percentage cover was used to convert existing dry weight values for algae species at the 24-month survey interval only. Dry weight values and algal species used can be found in Supplementary Material (Table 27).

Abiotic data were collected at 6-, 12-, 18- and 24-month intervals. A YSI multimeter was used for each rockpool to determine salinity (ppt) and temperature (Celsius). A ruler was used to measure water and sediment depth.

To test for statistically significant differences in percent cover, numeric counts, species richness, and between the rockpools and seawall over time, linear mixed effect models were run using the "nlme" package (Pinheiro et al. 2020) in R Studio (Version 1.2.1335). Only sessile species were used in statistical analysis to limit underestimation of species that photo-quadrats may have missed, such as mobile fauna. Assumptions of statistical tests were verified by examination of residuals against fitted model plots, as per Zuur et al. (2009). Where clear lack of normality or heteroskedasticity were identified, transformations were made to the data (log+1 transformations) before further statistical analysis. Habitat (rockpool interior, rockpool exterior and seawall), Level (top, middle, bottom) and Intervals (1-, 6-, 9-, 12-, 15-, 18-, 21- and 24-months) were fixed factors. To account for repeated measures within the rockpool and seawall, as the same rockpool and seawall photo-quadrats were surveyed each time, rockpool and photo-quadrat 'ID' was included as a random factor. To test for statistically significant differences in biomass between the habitats over different tidal levels using only the 24-month data, generalised linear models (GLM) were run with Habitat (rockpool interior, rockpool exterior and seawall) and Level (top, middle, bottom) as fixed factors. Quasi-Poisson distribution was used as data were over dispersed (Crawley 2012). Pairwise tests were run using the "emmeans" package (Lenth 2021).

Plymouth Routines in Multivariate Ecological Research (Primer-e v.7) was used to perform individual PERMANOVAs to test for differences in assemblage structure between Habitat, Level and Habitat * Level using sessile species abundance data from 24-month interval only (Anderson 2001). Data were square root transformed to avoid the weighting of common species over rare. A Bray–Curtis resemblance matrix was used with 9999 permutations and PERMANOVA run with unrestricted permutation of raw data. Significant results were followed by post hoc tests to determine if factors were significantly different. Multidimensional scaling (MDS) plots were used to visually demonstrate assemblage similarity between Habitat * Level.

6.4 Results

After two years in the intertidal environment, all rockpools were present and in good repair, despite significant storms in the intervening years since installation (Figure 30). On the middle and top seawall sections, it was possible to ascertain the presence of mobile fauna due to the very low abundance of canopy-forming seaweed. Only 1 *Littorina littorea* was recorded on the seawall. In and on the rockpools, 13 mobile species were recorded (Table 20).

Figure 30. The colonisation of column 4 at 6 months (a.), 12 months (b.), 18 months (c.) and 24 months (d.)

6.4.4 Abiotic

Salinity (ppt) ranged from 23.8 ‰ to 34.9 ‰ and was generally higher in the top rockpools. Main tests showed salinity did not significantly differ between rockpool levels, or level*interval, but it did between intervals (Table 19). Temperature ranged from 13.4 °C to 19.0 °C and was again generally slightly higher in the top rockpools (Figure 31). At the 6- and 18-month interval (both April), the temperature was significantly greater in the top rockpools compared to the middle and lower rockpools, but not at the 12- and 24-month intervals (both October).

Water depth ranged from 4.6 to 11.5 cm and varied between levels and between intervals, with no clear trend. Water depth in the middle rockpools was significantly greater than the top rockpools on two occasions, and significantly greater in the bottom rockpools than the top and the middle rockpools on one occasion. Water depth varied between rockpool levels the least at the final survey interval of 24 months. The maximum sediment depth recorded was 8.5 cm, and there is a positive temporal trend with sediment accretion increasing every interval for the bottom rockpools. Sediment accretion does not increase beyond 1 cm in the middle rockpools until 24 months, when the mean sediment height increases to 4 cm. Mean sediment height in the top rockpools does not exceed 1 cm throughout the study period. The bottom rockpool sediment depth is significantly greater than middle and top rockpool sediment depth for all intervals, with middle rockpool significantly greater than the top rockpool at 12- and 24-months.

Table 19. Main test results for abiotic factors salinity, temperature, water depth and sediment depth. Bold values indicate significant result.

Factor	numDF	denDF	F-value	p-value
Salinity				
tidal level	2	21	1.55	0.2366
survey interval	3	63	172.73	**<.0001**
level:interval	6	63	0.89	0.5095
Temperature				
tidal level	2	21	22.84	**<.0001**
survey interval	3	63	263.15	**<.0001**
level:interval	6	63	4.05	**0.0017**
Water Depth				
tidal level	2	21	3.051	0.0687
survey interval	3	63	9.611	**<.0001**
level:interval	6	63	3.031	**0.0114**
Sediment Depth				
tidal level	2	21	46.39332	**<.0001**
survey interval	3	63	58.74117	**<.0001**
level:interval	6	63	8.20152	**<.0001**

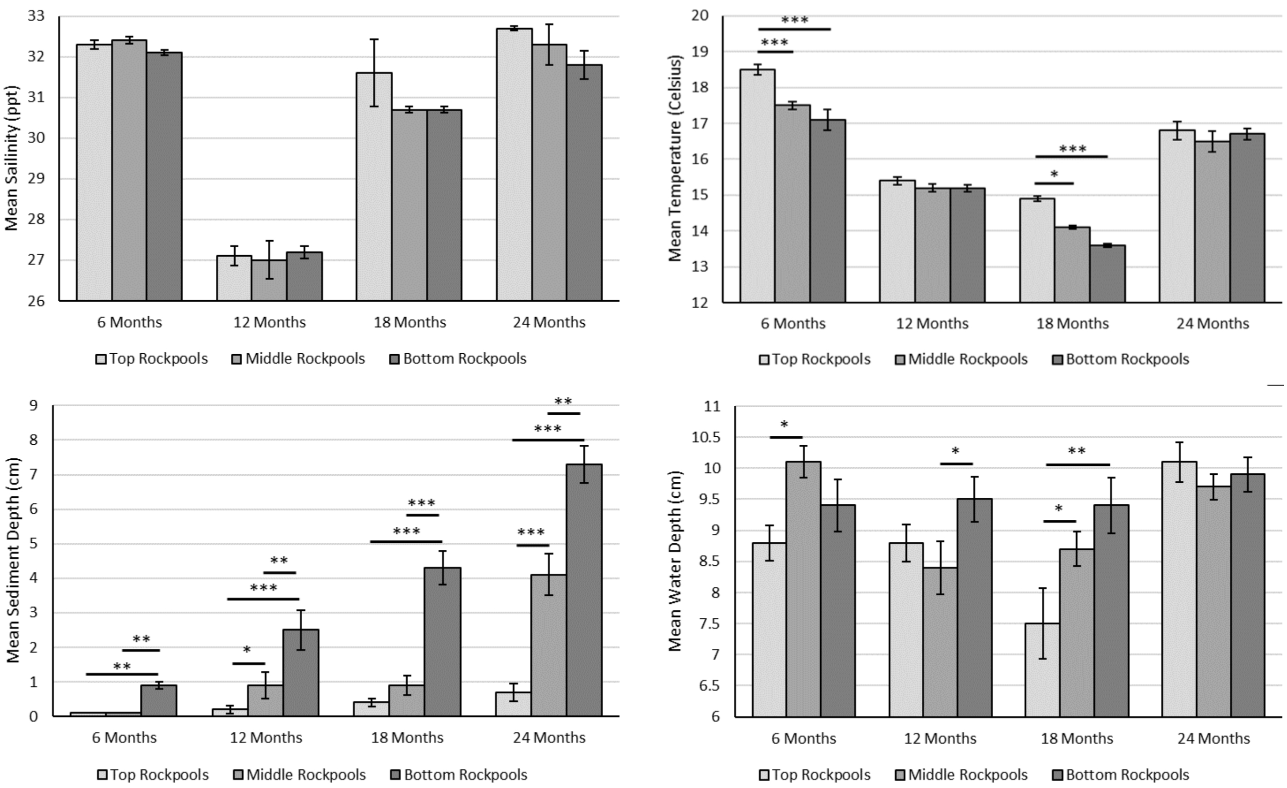


*Figure 31. Abiotic factors (top left, clockwise) salinity, temperature, water depth and sediment depth. Statistically significant differences indicated by * (<0.05), ** (<0.01) and *** (<0.001). Six, 12, 18 and 24 months correspond to April 2021, October 2021, April 2022 and October 2022 respectively. Error bars show standard error.*

6.4.5 Species richness

The total number of taxa recorded in the rockpools overall was 37, including mobile and sessile fauna and 3 species that were found in the sediment only. Thirty-one taxa were recorded in the rockpool interiors, and 16 on the rockpool exteriors. Two non-native species were identified; the barnacle *Austrominius modestus*, which occurred on both the seawall and the rockpools, and *Ficopomatus enigmaticus*, a calcareous tubeworm that occurred in the rockpools only (Table 20). Both species have been recorded elsewhere in the harbour (unpublished data). The climate migrant *Steromphala umbilicalis* was also recorded, which has only colonised this region of the UK coast within the past 20 years (Herbert 2023 personal communication). The top rockpool interiors were generally the most species poor, with species richness increasing inversely with the tidal level of rockpools. This trend was not reflected on the rockpool exteriors. Twenty-one sessile taxa were recorded in and on the rockpools, compared to 9 sessile taxa on the seawall.

Main tests showed significant results for all factors and interactions for both rockpool interior and seawall and rockpool exterior and seawall, except for habitat:level:interval for rockpool exterior and seawall (Table 21). Mean species richness peaked at 24 months for the rockpool interiors at all levels and at 18 months for the top and middle rockpool exteriors and 12 months for the bottom rockpool exteriors (Figure 32). From about 12 months onwards, the top and middle rockpool interiors and exteriors are significantly more species rich than the adjacent seawall.

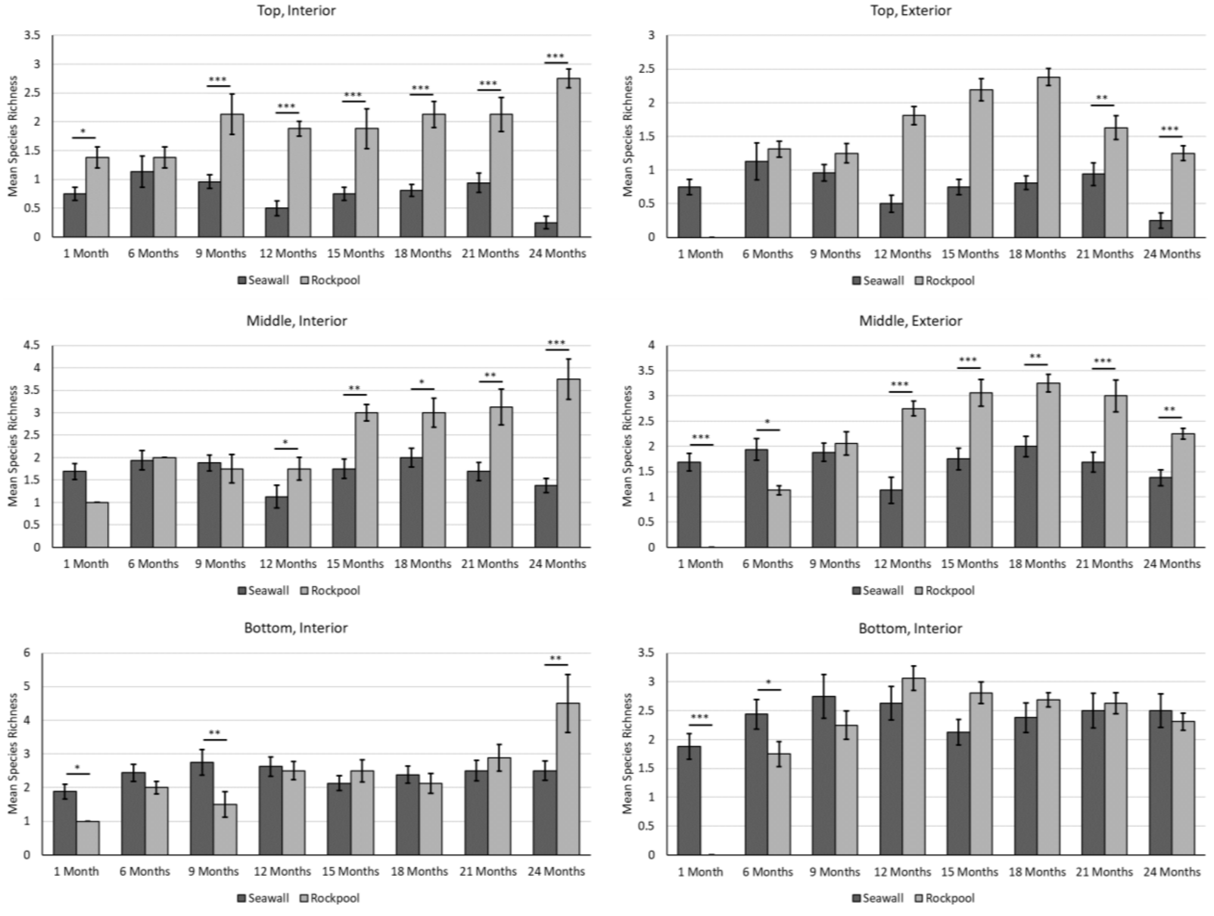
Table 20. Species recorded in the rockpool interiors, on the rockpool exteriors and on the seawall. Mean abundance given with standard deviation (±) given in brackets. Non-native taxa indicated with an asterisk (*). Taxa indicated with a dagger (†) denote those recorded as numeric counts, all other taxa were recorded as percentage cover.

	Rockpools Interior	**Rockpools Exterior**	**Seawall**
Brown Seaweeds			
Ascophyllum nodosum	0	0	15.1 (±28.4)
Fucus spiralis	5.7 (±16.6)	1.9 (±7.5)	1.5 (±5.7)
Fucus vesiculosus	6 (±20.6)	2.7 (±9.8)	4.5 (±13.5)
Pylaiella littoralis	14.6 (±23)	0.1 (±1.5)	0
Red Seaweeds			
Catanella sp.	0.01 (±0.07)	0	0.08 (±0.7)
Ceramium sp.	0.2 (±2.2)	0.01 (±0.09)	0
Dumontia contorta	0.02 (±0.1)	0	0
Porphyra sp.	0.01 (±0.1)	0.1 (±0.3)	0.003 (±0.05)
Polysiphonia sp.	0.1 (±0.7)	0	0
Green Seaweeds			
Blidingia minima	0	28.2 (±41.5)	16.7 (±27.2)
Cladophora sp.	2 (±11.9)	0	0
Diatoms	0.01	7.3 (±26.0)	0
Ulva spp.	11.2 (±18.4)	14.5 (±31.8)	0.2 (±1)
Porifera			
Halichondrea panacea	0.01 (±0.07)	0	0
Sycon ciliatum†	0.01 (±0.1)	0	0
Hydrozoa			
Clava multicornis	0.01 (±0.07)	0	0
Annelida			
*Ficopomatus enigmaticus**	0.1 (±0.4)	0	0
Spirobis spirobis	0.04 (±0.2)	0	0
Spirobranchus triqueter	0.1 (±0.2)	0	0
Crustacea			
Amphipoda sp. †	0.1 (±0.5)	0	0
Anurida maritima†	0.01 (±0.1)	0	0
*Austrominius modestus**	0.02 (±0.1)	1.1 (±4.3)	0.6 (±1.8)
Carcinus maenas†	0.3 (±0.6)	0.04 (±0.3)	0
Corophium volutator†	0.01 (±0.07)	0	0
Gammaridae sp.†	0.01 (±0.1)	0	0
Ligia oceanica†	0.1 (±0.3)	0.07 (±0.4)	0
Palaemon sp.†	1.6 (±3.3)	0	0
Semibalanus balanoides	0.01 (±0.07)	0.02 (±0.1)	0
Mollusca			
Hydrobidae sp.†	0.02 (±0.1)	0	0
Littorina littorea†	0.1 (±0.5)	0.04 (±0.2)	0.02 (0.2)
Littorina obtusata†	0.03 (±0.2)	0.02 (±0.2)	0
Littorina saxatalis†	0	0.01 (±0.07)	0
Mytilus edulis†	0.1 (±0.4)	0	0
Steromphala umbilicalis†	0.01 (±0.07)	0.01 (±0.07)	0
Vertebrata			
Lipophrys pholis†	0.02 (±0.2)	0	0
Total Taxa:	**32**	**16**	**9**

Table 21. Main test results for species richness for rockpool interiors and exteriors and seawall. Bold values indicate significant result.

Factor	numDF	denDF	F-value	p-value
Rockpool Exterior and Seawall				
habitat	1	66	6.3426	**0.0142**
tidal level	2	66	44.0404	**<.0001**
survey interval	7	654	94.0442	**<.0001**
habitat:level	2	66	7.2603	**0.0014**
habitat:interval	7	654	90.6078	**<.0001**
level:interval	14	654	5.2773	**<.0001**
habitat:level:interval	14	654	2.0513	**0.0127**
Rockpool Interior and Seawall				
habitat	1	66	24.4711	**<.0001**
tidal level	2	66	35.0013	**<.0001**
survey interval	7	462	7.9857	**<.0001**
habitat:level	2	66	10.9943	**0.0001**
habitat:interval	7	462	18.4841	**<.0001**
level:interval	14	462	4.2027	**<.0001**
habitat:level:interval	14	462	1.6539	0.0622

*Figure 32. Species richness for rockpool interiors and seawall (left column) and rockpool exteriors and seawall (right column). Mean species richness is per 25 x 25cm quadrat. Statistically significant differences indicated by * (<0.05), ** (<0.01) and *** (<0.001). One, 6, 9, 12, 15, 18, 21 and 24 months correspond to November 2020, April 2021, July 2021, October 2021, January 2022, April 2022, July 2022 and October 2022 respectively. Error bars show standard error.*



For the bottom rockpools, species richness is only significantly greater than the seawall in the rockpool interiors at 21 months. Seawall species richness is only significantly greater than rockpool species richness on seven occasions, and this predominantly occurs when comparing the seawall to the rockpool exterior during the first two survey intervals. Seawall species richness does not significantly exceed that of the rockpools after 9 months.

6.4.6 Abundance

Sessile percentage cover was highly variable throughout the study period with no linear trend observable (Figure 33). However, the pattern of abundance for both rockpool interior and exterior broadly follows a seasonal succession pattern, with initial colonisation of the rockpools dipping around the 12-month interval (October 2021) as boreal autumn and winter occurs. Percentage cover then increases from approximately 18 months (April 2022) onwards as new settlement and growth occurs, particularly perennial and slower growing species such as fucoid algae.

Main tests showed significant results for all factors and interactions for both rockpool interior and seawall and rockpool exterior and seawall, except for habitat for rockpool exterior and seawall (Table 22). For the bottom rockpools, abundance on the seawall is significantly higher at 9-, 12-, and 15-months than for the rockpool interior and at 1- and 15-months on the rockpool exterior. At no point is abundance within the rockpool interiors and exteriors higher than the seawall. However, abundance in the rockpools is significantly greater than the seawall for the middle and top rockpools, particularly from 21-months onwards.

Table 22. Main test results for percentage cover for rockpool interiors and exteriors and seawall. Bold values indicate significant result.

Factor	numDF	denDF	F-value	p-value
Rockpool Exterior and Seawall				
habitat	1	66	0.0018	0.9662
tidal level	2	66	20.3074	**<.0001**
survey interval	7	654	65.448	**<.0001**
habitat:level	2	66	4.6499	**0.0129**
habitat:interval	7	654	55.2333	**<.0001**
level:interval	14	654	9.5191	**<.0001**
habitat:level:interval	14	654	2.2834	**0.0047**
Rockpool Interior and Seawall				
habitat	1	66	4.4793	**0.0381**
tidal level	2	66	19.4243	**<.0001**
survey interval	7	462	9.6703	**<.0001**
habitat:level	2	66	6.6515	**0.0023**
habitat:interval	7	462	18.6644	**<.0001**
level:interval	14	462	3.0848	**0.0001**
habitat:level:interval	14	462	7.8868	**<.0001**

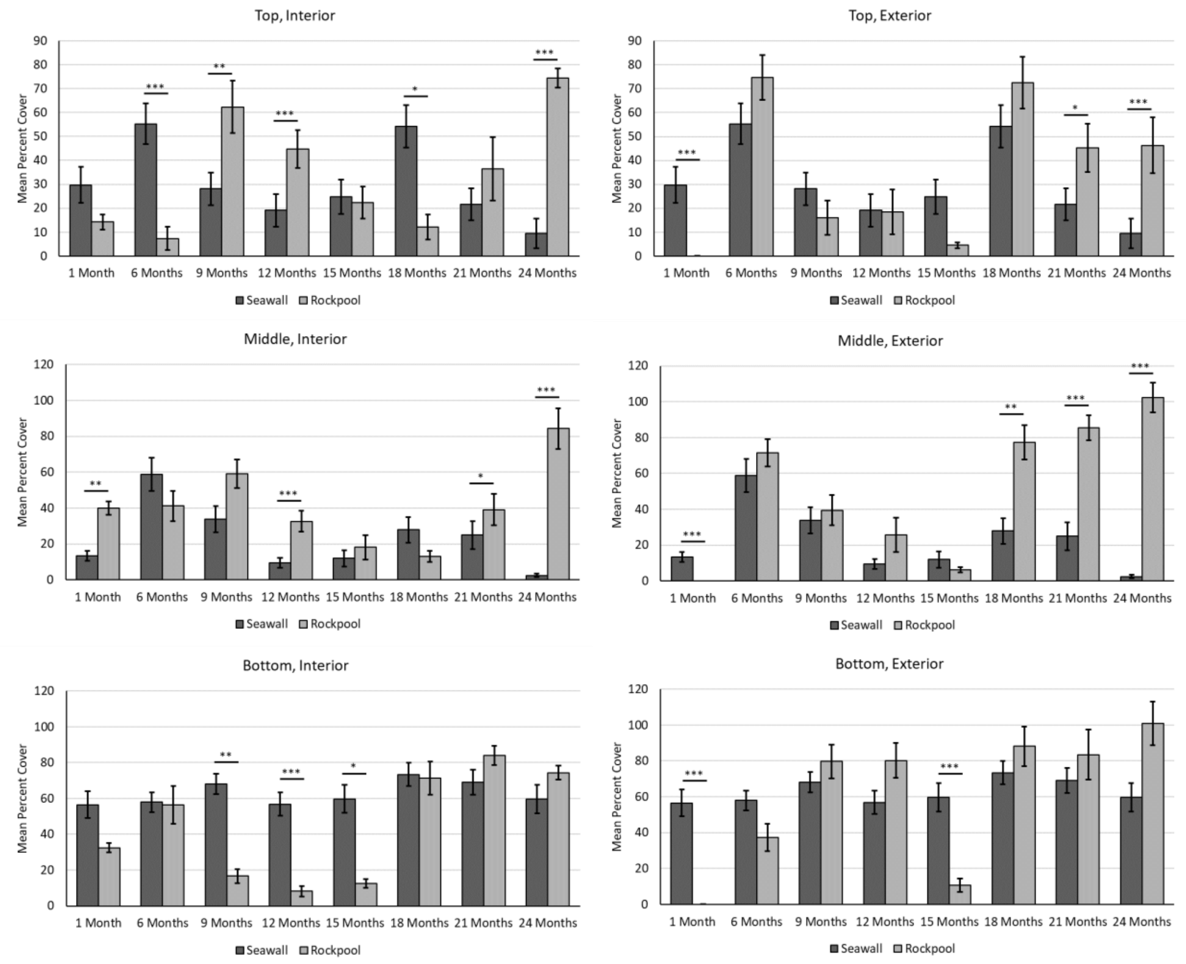


*Figure 33. Percent cover for rockpool interiors and seawall (left column) and rockpool exteriors and seawall (right column). Statistically significant differences indicated by * (<0.05), ** (<0.01) and *** (<0.001). Error bars show standard error.*

6.4.7 Assemblages

After 24-months, PERMANOVA main tests (Table 23) indicated that assemblage structures between both rockpool interiors and exteriors was significantly different to the seawall at tidal level, habitat and level*habitat. Post hoc tests (Table 24) show only the assemblages on the top rockpool exterior were not significantly different to the seawall and had an average similarity of 61.3%. Otherwise, all other results were significant (Table 24). Rockpool interiors and seawall were overall shared less average similarity than rockpool exteriors and the seawall. Average similarity between rockpool habitat and the adjacent seawall assemblages decreased linearly with tidal height. Grazing halos around the rockpools were observed (Figure 30) but the organism responsible was not identified.

Table 23. PERMANOVA main test results after 24 months for assemblages for rockpool interiors, exteriors and seawall. Bold values indicate significant result.

	Source	df	SS	MS	Pseudo-F	P (perm)
Rockpool Exterior and Seawall	Level	2	55950	27975	21.886	**0.0001**
	Habitat	1	47356	47356	37.048	**0.0001**
	Level * Habitat	2	24563	12282	9.6082	**0.0001**
Rockpool Interior and Seawall	Level	2	45464	22732	25.388	**0.0001**
	Habitat	1	57031	57031	63.693	**0.0001**
	Level * Habitat	2	27602	13801	15.413	**0.0001**

Table 24. PERMANOVA post-hoc test results after 24 months for assemblages for rockpool interiors, exteriors and seawall. Bold values indicate significant result.

Level	Groups	t	P (perm)	Unique perms	Average Similarity
Top	Exterior * Seawall	1.7809	0.0547	9901	61.3%
Middle	Exterior * Seawall	2.443	**0.0018**	9945	40.1%
Bottom	Exterior * Seawall	5.566	**0.0001**	9945	12.2%
Top	Interior * Seawall	9.7818	**0.0001**	9951	21.4%
Middle	Interior * Seawall	8.5799	**0.0001**	9941	19.3%
Bottom	Interior * Seawall	9.3761	**0.0001**	9943	17.7%

The MDS plots (Figure 34) for the rockpool interiors and seawall indicate that the assemblages of each habitat remain consistently divergent throughout the study period. The similarity between the rockpool interiors appears greater than the similarity between the seawall quadrats, even across tidal levels. By 24-months, the bottom rockpool interior assemblages appear most similar to the seawall, whereas the rockpool interiors of other tidal levels remain more distant from their seawall counterparts. However, the MDS plots for the rockpool exteriors show greater similarity to the seawall throughout.

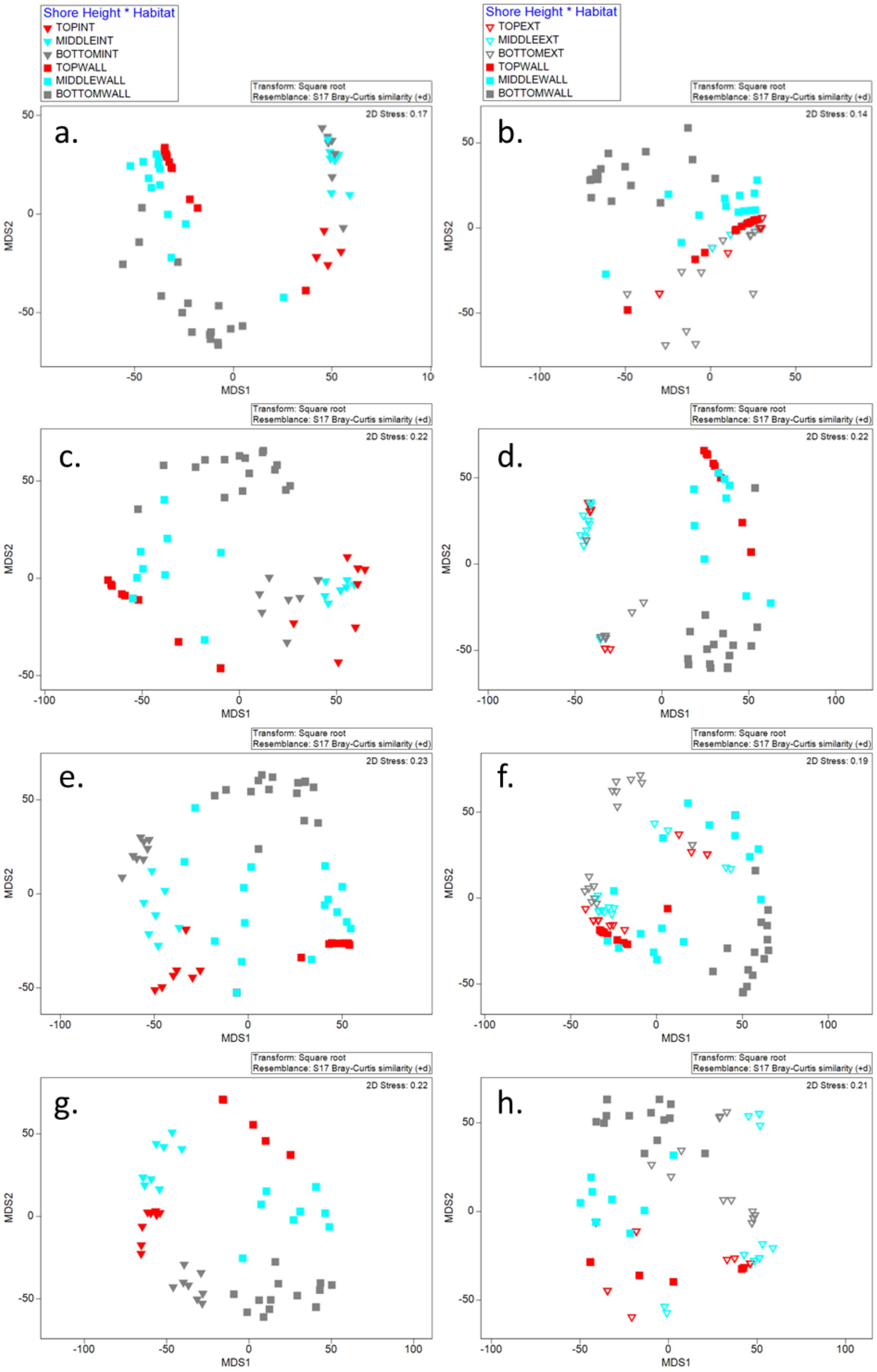


Figure 34. Multidimensional scaling plots indicating rockpool interiors and seawall (left column) and rockpool exteriors and seawall (right column), with 6 months (a., b.), 12 months (c., d.), 18 months (e., f.), and 24 months (g., h.) using species abundance data. Triangles correspond to rockpools and squares correspond to the seawall. Red indicates top level, blue indicates middle level and grey indicates bottom level.

6.4.8 Sediment infauna

Ten taxa were recorded in the rockpool sediment (Table 25) at 24 months. The species identified are fairly typical of an intertidal estuary, such as shore crabs *Carcinus maenas* and prawns *Palaemon* sp., but infaunal species, such as bivalves and polychaete worms, were rare. Mean species richness increased inversely with tidal height, with the most species occurring in the bottom rockpools (Figure 35). The sediment was mostly comprised of fine mud, with the occasional empty mussel shell or crab moult.

Table 25. Species recorded in the rockpool sediment at 24 months with mean abundance given.

	Top	Middle	Bottom
Annelida			
Hediste diversicolour	0.0	0.0	0.1
Polychaeta sp.	0.0	0.1	0.3
Crustacea			
Carcinus maenas	0.3	0.3	0.4
Palaemon sp.	0.5	2.1	5.3
Mollusca			
Bivalve	0.0	0.3	0.0
Hydrobidae sp.	4.6	1.4	9.4
Littorina littorea	0.0	0.3	0.3
Littorina obtusata	0.3	0.9	3.1
Mytilus edulis	0.0	0.0	0.1
Vertebrata			
Lipophrys pholis	0.0	0.0	0.1
Total Taxa:	**4**	**7**	**9**

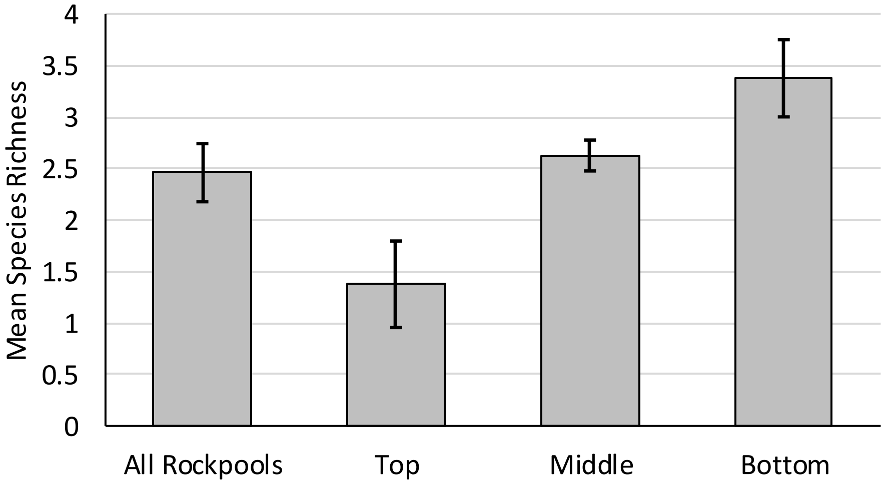


Figure 35. Mean species richness of rockpool sediment at 24 months (n=24). Error bars show standard error.

6.4.9 Biomass

After 24 months, main tests show significant differences for all factors for mean dry weight of sessile biota (Table 26), with tidal level contributing to variance the most (48.5%). The biomass on the rockpools significantly exceeds that on the seawall for middle level, but on the bottom level, biomass is significantly greater on the seawall (Figure 36). The mean dry weight of the bottom rockpools is the same as a single 25 x 25 cm quadrat on the adjacent seawall but the biomass of the seawall is more variable than that found on the rockpools. Mean biomass for the rockpool interior and exterior is relatively equal across tidal heights, each approximating half of the biomass for the whole rockpool.

Table 26. Main test results for biomass on the rockpools and seawall after 24 months. Bold values indicate significant result.

Factor	**Df**	**Deviance**	**Resid. Df**	**Resid. Dev**	***P***	**% Explained**
Biomass						
Habitat	2.0	118.8	93.0	1331.9	**<.0001**	8.2
Tidal level	2.0	703.0	91.0	628.8	**<.0001**	48.5
Habitat * Level	4.0	211.3	87.0	417.5	**<.0001**	14.6

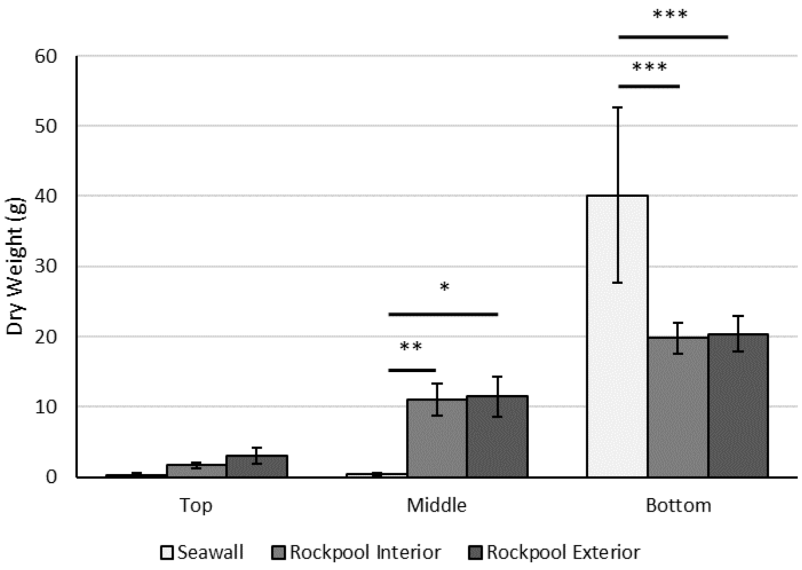


*Figure 36. Mean dry weight (g) of organisms for rockpool exterior and interior at 24 months. Statistically significant differences indicated by * (<0.05), ** (<0.01) and *** (<0.001). Error bars show standard error.*

6.5 Discussion

This study demonstrates that from mean tide level to high water neaps, artificial rockpools can enhance the species abundance of concrete seawalls and the magnitude of this impact is greater at higher tidal levels where refugia are more fundamental to survival. Chapman and Blockley (2009) found similar results, with the addition of pools on intertidal seawalls at lower shore levels less different to the seawall than those at mid and upper shore. The addition of a pool creates a new assemblage not found on the seawall in Hamble Harbour, in addition to sediment accretion that further provides habitat for a small number of infaunal taxa.

In the final surveys, species abundance for rockpool exterior and the seawall followed a similar pattern which was likely due to the high coverage of fucoid algae, predominantly *Fucus spiralis* and to a lesser extent *Fucus vesiculosus,* that developed on the rockpools. Species richness was greatest in the middle and bottom rockpools, which reflects trends in natural (Raffaelli and Hawkins 1996) and artificial rockpools (Firth et al. 2013). Further, species richness was significantly greater in the top rockpool interiors compared to the seawall just one month post instalment, which highlights the immediate benefits of refugia in the environmentally challenging artificial upper shore. These data suggest that artificial rockpools can supply ecological benefits across tidal levels and optimal placement of ecological enhancements is dependent on the desired outcome.

It is clear from these results that artificial rockpools can facilitate the vertical elevation of sessile organisms that are typically found in the lower shore or recorded only at the base of the seawall, which is reflective of the existing literature (Hall et al. 2019). The brown canopy-forming seaweed *A. nodosum* was restricted to the lower seawall, but was recorded on rockpools of all levels, though rarely on the top rockpools. The succession of opportunistic green algae to dense fucoid canopy on the middle and lower rockpools was also reported in Hall et al. (2019). The holdfasts of seaweeds *F. spiralis and F. vesiculosus* on the rockpools were observed almost exclusively on the rockpool rim. The horizontal, rough-textured rim surface likely provided an ideal attachment surface (Fletcher and Callow 1992) with good sunlight exposure, and this topographical pattern of fucoid colonisation has been observed on similar artificial rockpools elsewhere in the UK (Drakard et al. 2023). The incorporation of these features has facilitated the vertical migration of fucoid seaweed further up the tidal zone where it was otherwise absent.

The rockpools were also able to support more faunal species than the seawall, with a wide range of morphologies and traits, such as the delicate hydroid *Clava multicornis*. Most sessile faunal species were found exclusively in the rockpools, with only the non-native barnacle *Austrominius modestus* recorded on the seawall. Browne and Chapman (2014) found similar results, with only 2 sessile species on a vertical seawall in Sydney, but 7 in their bolt-on rockpools. Although mobile fauna was excluded from data analysis, they were abundant in the rockpools, particularly crabs *Carcinus maenas* and prawns *Palaemon* sp. Mobile organisms found in the rockpools, such as prawns and shanny *Lipophrys pholis*, would not survive on the seawall without water retentive features. The rockpools may also facilitate a halo effect (Fairweather 1988; Johnson et al. 1998), where grazers that would otherwise struggle to survive on a vertical seawall lacking microhabitats (Chapman 2006; Moreira 2006; Jackson et al. 2008) can find refuge at low tide and graze the immediate surrounding seawall, impacting the assemblage composition of the structure they're fixed to.

Sediment had accumulated in most of the middle and bottom rockpools by the end of the study, which has been recorded in other artificial rockpools (Firth et al. 2016; Waltham and Sheaves 2018; Bone et al. 2022). Previous work on a similar model artificial rockpool on the south UK coast has indicated that retained sediment can successfully host an infaunal assemblage comparable to that of a disturbed estuary (Bone et al. 2022), but that was not realised in this study. Although 3 infaunal taxa (*Hediste diversicolor*, unidentified polychaete, unidentified bivalve) were recorded, their abundance was rare, and the assemblage was instead dominated by crabs and prawns. The deposited mud was not particularly deep (≤ 7 cm) or compact, and so the low volume may have reduced its ability to provide a habitat analogous to a mudflat. The presence of crabs and fish in the sediment suggest it may get bioturbated by their movements and predation of infauna by crabs may limit their capacity to proliferate. However, the unintended retention of mud still plays a role as shelter, as crabs would bury themselves within the retained mud when disturbed during surveys.

6.5.1 Application

Sea levels are rising and will continue to rise (IPCC 2022), combining with coastal development and land reclamation (Dugan et al. 2011; Duarte et al. 2013; Duarte 2014) which results in coastal squeeze and intertidal habitat loss (Bugnot et al. 2021). Sea level rise scenarios (IPCC 2022) indicate that the top rockpools installed at high water neaps may in future be at lower tidal levels and will continue to deliver sufficient habitat for intertidal fauna where there previously was none. Spreading eco-engineering interventions across the vertical tidal zone will ensure that intertidal habitat remains available in the future, providing vital steppingstones to species' survival and ecological resilience. This also emphasises the requirement for interventions to be integrated into coastal development and engineering at the design and planning phase, as bolt-on interventions may not possess the required multidecadal longevity due to their often-protruding design and elevated risk of dislodgement. For example, Browne and Chapman (2014) lost several bolt-on 'flowerpot' artificial rockpools to wave action in Sydney Harbour, Australia.

To facilitate straightforward and accurate surveying, the rockpools in this study were devoid of macroscale features. In future, their design could be optimised by adding overhangs or ledges on the rim to create shaded areas, and a deeper pool to ameliorate extreme temperature values. Browne and Chapman (2014) found intertidal assemblages in rockpools with two different depths were not significantly different on the upper shore, and so the impact of rockpool depth may not be as great on upper shore levels. However, design choices can be made to meet a variety of ecological needs and engineering standards. Consultation and collaboration with the appropriate experts are fundamental to implementing successful and climate resilient eco-engineering interventions.

6.6 Conclusion

This study has demonstrated that artificial rockpools can provide valuable hard and soft substrate habitats between high water neaps and mean tide level on a concrete seawall by retaining water and sediment. The rockpools provided crucial refugia to a wide range of species that were not otherwise present on the seawall and extended the vertical elevation of habitat-forming species such as canopy-forming brown seaweed. Installation of rockpool interventions should be incorporated across the tidal zone, including the upper shore where their presence will be of greater importance in the coming decades due to climate-change induced sea level rise and warming. Retrofitted, bolt-on rockpools have adequately demonstrated proof of concept and so future interventions should be integrated into the initial design phase of coastal infrastructure, negating the requirement for retrofitting.

6.7 References

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6.8 Supplementary material

Table 27 The dry biomass reference values used to convert percentage cover into biomass. Biomass values were obtained by drying algae at 100 °C until a constant weight was achieved. Algae samples were obtained from another harbour on the south coast of England in the spring and summer of 2022.

Seaweed taxa	**Biomass (g) per 1% cover**
Ascophyllum nodosum	0.898
Pylaeilla sp.	0.013
Fucus spiralis	0.230
Fucus vesiculosus	0.287
Catanella sp.	0.436
Ceramium sp.	0.020
Polysiphonia sp.	0.012
Blidingia minima	0.034
Cladophora sp.	0.046
Ulva spp.	0.022

Chapter 7 – Estuarine infauna within incidentally retained sediment in artificial rockpools

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7.1 Abstract

Artificial coastal structures (ACSs) are primarily designed to provide services for human use, such as flood defence or shipping, and are generally poor for marine biodiversity. Consequently, there has been significant research effort to enhance these hard structures to increase biodiversity and habitat availability via eco-engineering. On seawalls and breakwaters, this has included the creation of habitats for benthic species found on natural rocky shores, including the provision of cracks, crevices and water retaining features, such as artificial rockpools. When sediment retention in these features has occurred, it has often been deemed detrimental to the overarching aim of the intervention. Yet, it is soft sediment habitat that is impacted the most through coastal construction.

As ecological enhancement of a flood defence scheme, nine concrete retrofit rockpools were installed at three different tidal elevations between mean high water neap tide and mean tide level on steel sheet piling on the Arun Estuary in Littlehampton Harbour, UK, which naturally filled with mud one year after installation. To explore how analogous the faunal assemblages and sediment profile of rockpool mud were to two local mudflats, core samples were taken and analysed for species richness, abundance, biomass, assemblage structure, median grain size, and organic matter content.

More benthic species were observed in the artificial rockpool than in the local mudflats. Although the rockpools were placed at higher tidal levels than the lower shore mudflat, their assemblage structure and species richness were more similar to the lower shore mudflat at the base of the sheet piling than the upper shore mudflat. This study demonstrates that retained sediment within eco-engineered features on hard ACSs can create habitat for benthic assemblages. Providing sediment-retentive features on ACSs has the potential to provide a novel eco-engineering option that may be appropriate for some heavily modified waterbodies on sheltered, depositional coasts.

7.2 Introduction

Coastal squeeze threatens developed coasts with habitat loss as the intertidal area is reduced and steepened between a fixed high tide mark on artificial coastal structures (ACSs) and sea level rise (Doody 2004; Schleupner 2008; Pontee 2011) which is predicted to significantly decrease biodiversity within coastal regions (Hawkins 2012; Hawkins et al. 2016). ACSs can physically reduce soft-sediment habitat from their construction footprint (Bugnot et al. 2021) and may also contribute to the alteration of natural dynamic soft-bottomed and hydrological processes (Dugan et al. 2018) and the subsequent impacts on benthic species diversity and community structure (Martin et al. 2005; Heery et al. 2017; Critchley and Bishop 2019).

It is known that ACSs are not analogous habitats to natural intertidal reefs in terms of habitat availability and topographical complexity (Moschella et al. 2005; Chapman and Underwood 2011; Aguilera et al. 2014) as well as community structure, species richness and biodiversity (Connell and Glasby 1999; Chapman 2003; Moschella et al. 2005; McKinney 2006; Glasby et al. 2007; Vaselli et al. 2008; Pister 2009). Consequently, there has been significant effort to increase biodiversity and habitat availability via eco-engineering (Firth et al. 2014; Strain et al. 2017; O'Shaughnessy et al. 2019), where habitat is integrated into ACSs (Bergen et al. 2001; Mitsch and Jorgensen 2003; Odum and Odum 2003). The predominant aims of eco-engineering on ACSs thus far have been to increase topographical and structural complexity (MacArthur et al. 2019), either via texture (Coombes et al. 2015) or through the creation of microhabitats (Martins et al. 2010; Martins et al. 2016; Loke et al. 2017), and moisture retentive features, such as drill-cored tide pools and retrofitted artificial rockpools (Browne and Chapman 2014; Hall et al. 2019; Chee et al. 2020).

To date, eco-engineering has focused exclusively on mitigating habitat loss of natural intertidal reefs with the addition of interventions to retain seawater for biota that would normally occur on hard substrate. Yet, it is soft-sediment habitat that is often impacted the most through construction (Airoldi et al. 2005; Firth et al. 2016), from the seabed offshore (Miller et al. 2013) to the coastal benthos (Martin et al. 2005; Bulleri and Chapman 2010). When sedimentation in eco-engineering features has occurred, it is deemed incidental and potentially detrimental to the overarching aim of the design (*sensu* Hall et al. 2018; Waltham and Sheaves 2018). However, in natural ecosystems, sediment provides a substrate for infauna and potential ecosystem services (Barbier et al. 2011; Costanza et al. 2014; Dissanayake et al. 2018), such as organic matter sequestration and nutrient recycling (Cook et al. 2004a, b, Watson et al. 2020). At the time of writing, there were no published studies that quantified the biotic assemblages inhabiting sediment accumulated in eco-engineering installations, either by design or incidentally.

Coastal structures disproportionately impact the species richness of muddy sediments compared to sandy sediments, particularly the abundance of infauna such as nereid worms and bivalves (Critchley and Bishop 2019). Many of these species contribute to valuable ecosystem processes and services. For example, the ragworm *Hediste diversicolor* (O.F. Müller, 1776) is important prey for birds and fish (McLusky and Elliott 2004) and bioturbation caused by burrowing activity is known to influence biogeochemical cycles (Davey and Watson, 1995; Gunnarsson et al. 1999; Garcia-Arberas and Rallo 2002). Sediment characteristics, such as total organic matter and median grain size, can be determining factors in species richness and community structure of intertidal infauna (Ellingsen 2002; Ysebaert and Herman 2002; Coblentz et al. 2014; Coblentz et al. 2015). Species size distribution can be indicative of effective recruitment and post-recruitment survival, which is important for maintenance of populations (Hunt and Scheibling 1997; Dethier et al. 2012; Beal et al. 2018).

Muddy sediments are more likely to occur in sheltered locations, such as estuaries and harbours, where additional stressors co-occur such as greater nutrient loading (Jordan et al. 2018). The installation of eco-engineered features on ACSs, such as on harbour walls and wharves, in these areas are more likely to retain sediment. Therefore, understanding how soft-sediment assemblages compare with natural mudflat habitats is beneficial. In 2018, nine eco-engineered artificial rockpools (known commercially as Vertipools™), were installed on steel sheet piling in Littlehampton Harbour, West Sussex, UK. It is one of the first known examples of this type of intervention retrofitted to steel coastal structures. The intention was to add complexity to the harbour wall constructed from steel sheet piling and to provide maximum opportunity for colonisation, with the artificial rockpools known to retain mostly seawater at low tide (*sensu* Hall et al. 2019) and become colonised by benthos characteristic of intertidal rocky shores. However, all artificial rockpools filled to the brim with estuarine mud. To determine what faunal assemblages were found in the rockpool mud and to compare how analogous these artificial habitats were compared to existing local mudflats, sediment samples were taken from both the artificial rockpools and two local mudflat sites. The main research question was, how do biotic assemblages and the sediment characteristics of the artificial rockpool mud compare to local mudflats?

7.3 Method and materials

7.3.1. Study site

Littlehampton Harbour is a heavily modified waterbody situated on the south-east coast of England (50.809596, -0.54879069) at the mouth of the River Arun (Figure 37). It is a small, busy port popular with recreational boating. Its eastern shore is predominantly comprised of steel sheet piling and floating pontoons, with a greater presence of gently sloping mudflats on the western shore. At low water, intertidal mudflats are exposed, and vessels are limited to a narrow channel in the centre of the river. The harbour entrance has a southerly aspect, which is sheltered from prevailing south-westerly winds by West Beach and the presence of a breakwater that juts out past the river mouth. Tidal streams are exceptionally strong reaching 6 knots (3 m s^{-1}) with ebb flows increased after heavy rain. The study site is approximately 1 km north of the harbour entrance. Mean spring tidal range is 5.5 m.

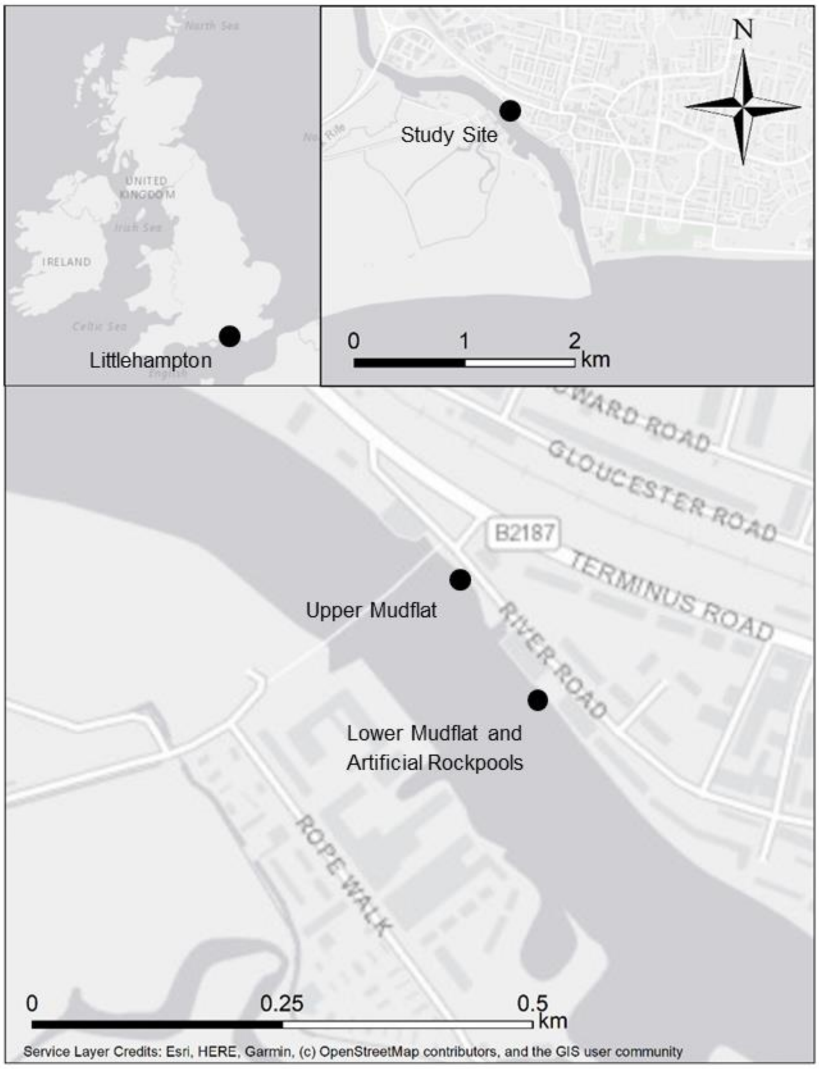

Figure 37. Location of survey sites at Littlehampton Harbour on the River Arun, south coast of England.

7.3.2 Artificial rockpools

As part of flood defence improvements between 2013 and 2018, new vertical steel sheet pile flood defence walls, capped with concrete coping, were installed along the Eastern Bank of Littlehampton Harbour (Figure 38). A section of this sheet piling included the installation of nine artificial concrete rockpools arranged in three columns of three rockpools at different tidal heights. All rockpools were fixed within the 'in-pans' (concave inlets in the structure) of the steel sheet piling (Figure 38). The lowest rockpool in all columns was 40 cm from the toe (± 10 cm) with approximately 50 cm height between each rockpool. The top artificial rockpools were fixed to a height approximating to Mean High Water Neap Tide Level (Table 28). The artificial rockpools were commissioned by the Environment Agency and manufactured by Artecology Ltd. (Sandown, United Kingdom), based on their Vertipool™ Compton design. They were fitted to the steel sheet piling in 2018 as it was constructed, with a custom stainless-steel bracket inlaid into the low-carbon concrete artificial rockpools. The dimensions of the artificial rockpools were 15 cm (h) x 31 cm (w) x 25 cm (d). Maximum mud depth of rockpool interior when full was 6 cm. The experimental design of this study was constrained by the number (n = 9) and arrangement of the artificial rockpools as they were not installed for research purposes.

Figure 38. Left: One of the three columns of artificial rockpools. The rockpools were fixed to the in-pan of the steel sheet piling. The top rockpool was fixed at approximately Mean High Water Neap Tide Level. The foot of the sheet piling was approximately Mean Tide Level. Right: an artificial rockpool on the steel sheet piling filled with mud, photographed 2.5 years after installation.

Table 28. Tidal height and number of core samples taken from artificial rockpools and mudflat habitats. The top rockpool was at approximately Mean High Water Neap Tide Level. The foot of the sheet piling was approximately Mean Tide Level.

Habitat	Height Above Chart Datum (m)	Number of Sampling Stations	Number of Core Samples
Artificial Rockpools	3.25 - 4.25	9	27
Upper Mudflat	3.25 – 4.25	9	27
Lower Mudflat	2.75	9	27

7.3.3 Survey method

The artificial rockpools and mudflats were both sampled on the same day in September 2020, two and a half years after installation, commencing on an ebbing tide 1 hour prior to low water when all habitats were emersed, and finishing 2 hours post low water. Three sediment cores were taken from each artificial rockpool using a 6 cm diameter corer to a depth of 5 cm. The core size, sampling depth and number of cores was constrained by the size of the artificial rockpool. A single core was taken and retained in an airtight re-sealable bag for sediment analysis and stored in a freezer (-18 °C) the same day to prevent decomposition of organic material. Two additional cores from each artificial rockpool were combined and sieved on site using a 0.5 mm sieve to retain macrofauna, which were transferred to tubs and preserved with 10 % formal saline (Fisher Chemical, Loughborough, United Kingdom). The combined cores from each rockpool total an area of 0.075 m^2.

The lower mudflat below the artificial rockpools and an upper mudflat 100 m northwest (Figure 37) were both sampled as reference sites. These are also referred to as 'natural' mudflats, however as the estuary has been highly modified, these are potentially contaminated, and the benthic assemblages may differ to mudflats at unmodified locations. Alternative and additional reference sites within the harbour were inaccessible, and sites further away presented the issue of confounding variability. The lower mudflat at the toe of the sheet piling was sampled due to its close proximity but is at a lower tidal height (approximately Mean Tide Level) than the artificial rockpools and upper mudflat. Core samples were also taken at the approximate tidal heights (Mean High Water Neap Tide Level) of the artificial rockpools (± 15 cm) at an upper shore mudflat. Due to the sheet piling restricting access to upper shore mudflats, the closest upper mudflat was approximately 100 m upstream where the sheet piling terminated (Figure 37). Three cores (6 cm diameter, 5 cm depth) were taken randomly at each of nine sampling stations on the lower mudflat at the toe of the sheet piling (n=9) and at nine sampling stations on the upper mudflat at a shore height equivalent to the artificial rockpools (n=9). To replicate the sampling method used for the artificial rockpools, one core was taken for sediment analysis, and two cores were taken and combined for faunal analysis at each of the nine sampling stations for both mudflats (Table 28).

The sediment samples were defrosted, homogenised and subsampled for sediment analysis. Organic content was determined by drying a homogenised subsample in a Memmert UN55 drying oven (Memmert, Büchenbach, Germany) at 100 ℃ for 48 h, and then placing in a Carbolite Gero CWF chamber furnace (Carbolite Co Ltd., Sheffield, United Kingdom) at 450 ℃ for 12 h and measuring the loss of mass on ignition (% LOI) (Luczak et al. 1997). Water content was determined as the loss of weight following drying. Particle size analysis was conducted using a Mastersizer 3000 laser diffractometer (Malvern Panalytical, Malvern, United Kingdom) with subsamples of the dried sediment. Each sample produced five median grain size (D_{50} – sieve opening which 50% of particles pass through (Yilmaz et al. 2019) readings which were used in data analysis (Shakeel et al. 2020).

The biotic samples were rinsed of preservative and viewed under a Leica M165C stereo microscope (Leica, Wetzlar, Germany). Fauna were picked and placed in tubes containing 70% Industrial Methylated Spirit (IMS) (Fisher Chemical, Loughborough, United Kingdom) for longer term storage as voucher specimens. Identification of individuals was made using appropriate keys to lowest taxonomic resolution. Organisms without a head and empty shells were discounted. Biomass was obtained by drying organisms in a Memmert UN55 drying oven at 100 °C until a constant weight was achieved. Species with shells were left intact as specimens were too small to remove tissue manually. To ensure preservation did not cause changes in weight, organisms were weighed immediately upon removal from preservation fluid to reduce exposure times. The size-frequency of ragworm *H. diversicolor* was determined in each habitat to compare recruitment potential. The total length of all sampled ragworms were measured using a BMZ-07 stage micrometer calibrated with a BMZ-06 x10 measuring eyepiece graticule (Brunel Microscopes Ltd., Chippenham, United Kingdom). The peristomium width and total body length from the prostomium to pygidium (when entire specimen was present) was measured. Although metric L3 (sum of the lengths of the prostomium, peristomium and the first chaetiger) is often preferred operationally in condition assessments, Total Length was consistently highly correlated with wet weight (Durou et al. 2007) and considered appropriate for this comparative investigation. Body length data for incomplete specimens was extrapolated from a linear trendline of peristomium width and total length plotted on a scatter graph of complete specimens.

7.3.4 Data analysis

Data were divided into their respective habitats: artificial rockpool, lower mudflat and upper mudflat. For comparison with the natural mudflats, data from the nine rockpools were treated as a single sample set to compare with the nine sampling stations for each mudflat habitat. Due to low replication (n=3), analysis of tidal height was not performed on rockpool data. Plymouth Routines in Multivariate Ecological Research (PRIMER-e, version 1.0.6) was used to detect similarities in assemblages between the habitats (Clarke and Gorely 2006). Species abundance data were square root transformed, to prevent common species being weighted over rare species, and a Bray-Curtis similarity matrix was created between samples. A single factor PERMANOVA design (habitat: artificial rockpool, lower mudflat, upper mudflat) was used to assess the difference in assemblages using species abundance data (Anderson 2001). Multidimensional scaling (MDS) plots were produced to visually demonstrate the similarity between sample assemblages for habitats and MVDISP was used to provide dispersion indices. SIMPER was used to identify where dissimilarity occurred between habitats and which taxa were most responsible for these differences in assemblage structure. The DIVERSITY function was used to calculate Shannon Weiner diversity index (Clarke and Gorely 2006).

R version 3.6.3 (R Core Team, 2016) and the 'car' (Fox and Weisberg 2019) and 'userfriendlyscience' (Peters 2016) packages were used to run one-way analysis of variance (ANOVA) tests for average moisture content, organic matter content, median grain size, species richness, abundance, and total biomass with habitat as a fixed factor. Data were tested for normality with Shapiro Wilkes test and were square root transformed if p=<0.05. Tukey *post hoc* tests were run to determine the direction of significant interactions as indicated by the one-way ANOVAs. A Levene's test identified unequal variances (p=<0.05) in Shannon Weiner diversity indices, and so a Welch's ANOVA and Games Howell post hoc test were run instead. ANOVA results are presented as degrees of freedom (df), the F statistic (F), and the p value (p).

7.4 Results

7.4.1 Sediment profile analysis

Artificial rockpool samples were visibly wetter. A one-way ANOVA revealed the average moisture content (% of weight) of the habitats was significantly different (df 2,23, F=13.05, p=<0.001). Pairwise tests showed moisture content of artificial rockpool sediment (39 %) was significantly greater than both lower (p=<0.01) and upper (p=<0.001) mudflats (Figure 39). The organic content (% loss on ignition, Figure 39) was also significantly different (one-way ANOVA, df 2, 23, F=6.579, p=<0.01) between habitats and was significantly higher in the artificial rockpools than the lower mudflat (p=<0.01). The one-way ANOVA for median grain size showed a significant difference between median grain sizes of habitats (df 2, 24, F=350.09, p=<0.001). Median grain size was significantly smaller in the rockpool sediment compared to both mudflats (p=<0.001). Mean median grain size ranges between 48 µm for rockpool sediment and 56µm for lower mudflat sediment. All habitat samples are predominantly comprised of silt/ mud (≤63 µm) (Wentworth 1922).

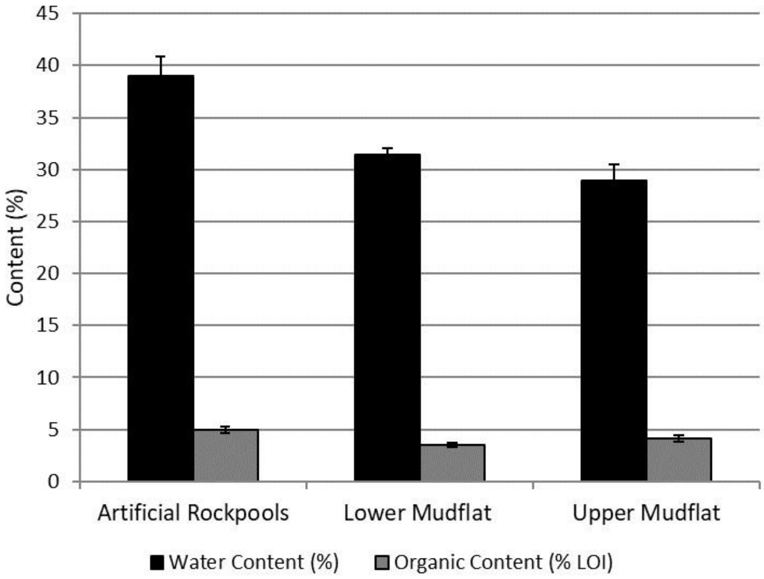

Figure 39. Mean water and organic content in the soft-bottomed habitats expressed as percentage. Error bars show standard error (n=9).

7.4.2 Faunal analysis

A total of thirteen taxa were identified across all habitats, with the gastropod snail *Peringia ulvae* numerically abundant in all habitats. The presence of other taxa was variable depending on the habitat (Table 29). No non-native species were identified in the sediment. However, the non-native barnacle *Austrominius modestus* was identified on the rockpool exterior and was also present on the sheet piling.

The artificial rockpool habitat contained eleven taxa, the highest for all habitats, with three taxa found only in the artificial rockpool sediment in very low abundances: Actiniidea indet., *Cirratulidae* sp. and *Corophium volutator*. Annelid worms were the most abundant taxonomic group. Mean species richness was five (Figure 40) and ranged between four and seven. Specimens of the bivalve *Scrobicularia plana*, found in all habitats, were all juveniles (<15 mm shell length). The lower mudflat habitat contained seven taxa with one species found only within this habitat: the polychaete *Capitella capitata*. Species richness ranged from five to seven species in samples and the lower mudflat had the highest mean species richness of all habitats with 6.4 species (Figure 40). The upper mudflat habitat contained eight species of which the gastropod *P. ulvae* was the most numerically abundant. Annelid worms were absent from most sample stations from the upper mudflat. Mean species richness was the lowest of all habitats with 3.2 species (Figure 40).

Table 29. Species list for the habitats showing mean density (m²). Asterisk indicates where taxa were found only in that habitat. Note that samples were taken in the upper 5 cm of mud profile only. Standard error given in brackets (±).

	Artificial Rockpool	**Lower Mudflat**	**Upper Mudflat**
Phylum Cnidaria			
Anthozoa			
Actiniidae *indet.*	20* (20)	0	0
Phylum Annelida			
Capitella capitata	0	432* (184)	0
Cirratulidae sp.	20* (20)	0	0
Hediste diversicolor	3007 (938)	3223 (543)	197 (123)
Streblospio shrubsolii	1395 (577))	8843 (1655)	0
Tubificoides sp.	570 (171)	2456 (272)	20 (20)
Phylum Arthropoda			
Isopoda			
Cyathura carinata	39 (39)	1159 (272)	20 (20)
Amphipoda			
Corophium volutator	39* (26)	0	0
Decapoda			
Carcinus maenas	59 (30)	0	20 (20)
Insecta			
Coleoptera sp.	0	0	20* (20)
Dolichopodidae sp.	59 (59)	0	373 (116)
Phylum Mollusca			
Gastropoda			
Peringia ulvae	3675 (955)	3007 (526)	6328 (1901)
Bivalvia			
Scrobicularia plana	98 (43)	314 (97)	275 (73)
Total Species Richness	**11**	**7**	**8**

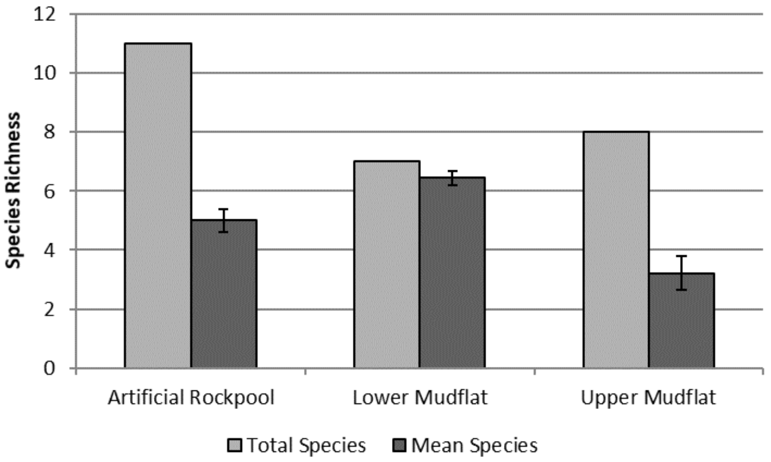

Figure 40. Species richness for soft-bottomed habitats. 'Total species' indicates the number of different taxa sampled in a habitat, whereas 'mean species' is the average number of species per sample station/ rockpool within that habitat (n=9). Error bars show standard error.

The one-way ANOVA showed a significant difference between habitats for species richness (df 2,24, F=14.89, p=<0.001) and abundance (df 2,24, F=8.08, p=<0.01). Tukey *post hoc* tests (Table S31 Supplementary Material) revealed significant differences between upper mudflat-lower mudflat (p=<0.001) and upper mudflat-rockpools (p=<0.05) for species richness, and between upper mudflat-lower mudflat (p=<0.01) and rockpools-lower mudflat (p=<0.05) for abundance.

The upper mudflat was least diverse with a mean Shannon Weiner diversity index (H') of 0.67 (± 0.11 standard error) and showed the greatest range (0 – 0.89). The lower mudflat ranked most diverse (H'=1.38 ±0.06 standard error) and showed the smallest range (1.01-1.54). The artificial rockpools ranked in the middle (H'=1.17 ±0.08 standard error) but their diversity indices were closest to the lower mudflat. A Welch's ANOVA demonstrated significant difference between the diversity indices of the different habitats (F=11.207, p=<0.01) and a Games Howell post hoc test revealed significant differences between upper mudflat-lower mudflat (p=<0.01) and upper mudflat-rockpools (p=<0.01).

The lower mudflat sediment contained the greatest total biomass (159 g m^{-2}), with a mean biomass of 18 g m^{-2}, dominated by *H. diversicolor* (Figure 41). Artificial rockpool sediment closely followed with a total and mean biomass of 134 g m^{-2} and 15 g m^{-2} respectively, dominated again by *H. diversicolor*. Upper mudflat sediment contained the least total and mean biomass with 55 g m^{-2} and 6 g m^{-2} respectively, dominated by gastropods. In samples where ragworms were present, biomass was likely to be underestimated due to 38 % of *H. diversicolor* specimens appearing incomplete. Hence, the biomass results should be interpreted with caution. One-way ANOVAs revealed there was no significant difference (df 2,24, F=2.36, p=>0.116) between the total biomass of the different habitats. The lower mudflat and artificial rockpool mud assemblages were most similar to the littoral sediment biotope '*H. diversicolor* and *Streblospio shrubsolii* (Buchanan, 1890) in littoral sandy mud' (LS.LMu.UEst.Hed.Str) (Connor et al. 2004).

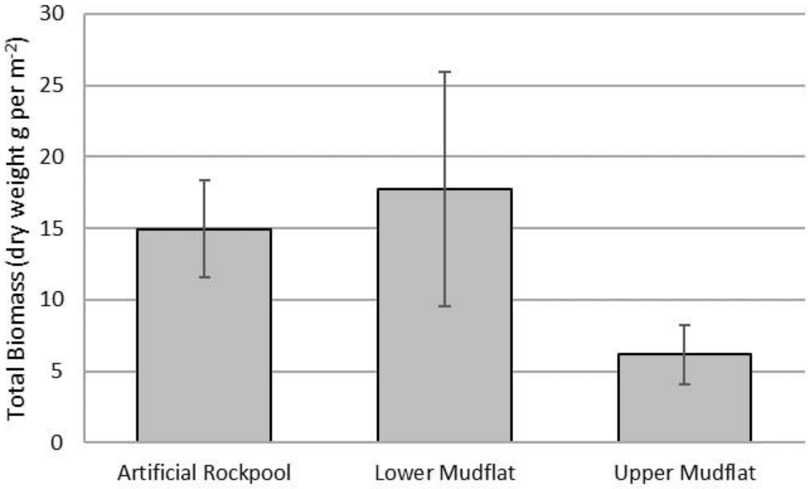

Figure 41. Mean total biomass for soft-bottomed habitats. Error bars show standard error.

The *H. diversicolor* size class with the highest proportion for the artificial rockpools was 11-20 mm (Figure 42). The lower mudflat had a greater proportion of the smallest size class (1-10 mm) than the artificial rockpools, but both were predominantly comprised of *H. diversicolor* between 1 and 30 mm. Artificial rockpools had a slightly higher proportion of the larger size classes than the lower mudflat. The upper mudflat had very low abundances of *H. diversicolor* and their sizes did not exceed 30 mm.

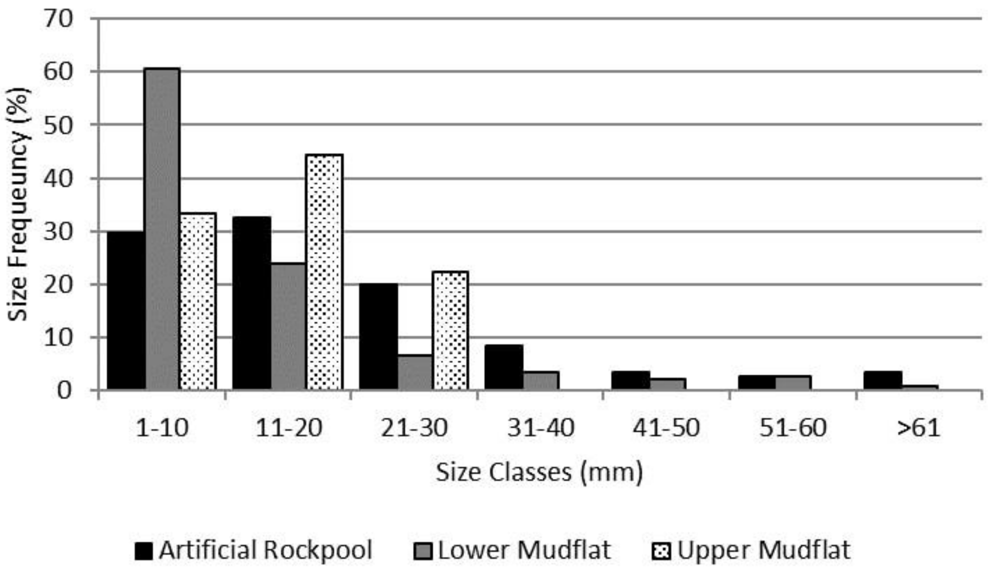

Figure 42. Size frequency (total length) of the ragworm Hediste diversicolor in the artificial rockpools (n=153), lower mudflat (n=164), and upper mudflat (n=10).

PERMANOVA pairwise tests (number of permutations = 9999) revealed significant differences in species abundance data between all habitats (p=<0.001) (Table 30).

Table 30. Pairwise tests for PERMANOVA between soft-bottomed habitats on species abundance data (Number of permutations = 9999).

Groups	t	P(perm)	Unique perms
Artificial rockpool, Lower Mudflat	2.6616	0.0004	8177
Artificial rockpool, Upper Mudflat	2.9279	0.0001	8123
Lower Mudflat, Upper Mudflat	5.1657	0.0002	8179

MDS plots of the soft-bottomed habitat species abundance data showed the habitat plots as relatively discrete groupings with little overlap (Figure 43). The lower mudflats plots are tightly clustered, which was reflected by a low dispersion index (0.568). The artificial rockpool and upper mudflat plots, however, were more dispersed (dispersion indices are 1.281 and 1.151 respectively); indicating within-habitat artificial rockpool and upper mudflat samples were not as similar as within-habitat samples of the lower mudflat. This was reflected by the pairwise PERMANOVA, which showed that the average similarity within the habitats was highest for the lower mudflat (77.4) and upper mudflat (60.0) and lowest for the artificial rockpool (56.0). The MDS artificial rockpool plots sat between both the mudflat habitats but were plotted slightly closer to the lower mudflat plots, indicating a greater similarity than with the upper mudflat plots. Similarity between artificial rockpools and lower mudflat was reflected by the pairwise test, which indicated average similarity of 54.6 between artificial rockpool and lower mudflat assemblages and 40.7 between artificial rockpool and upper mudflat assemblages.

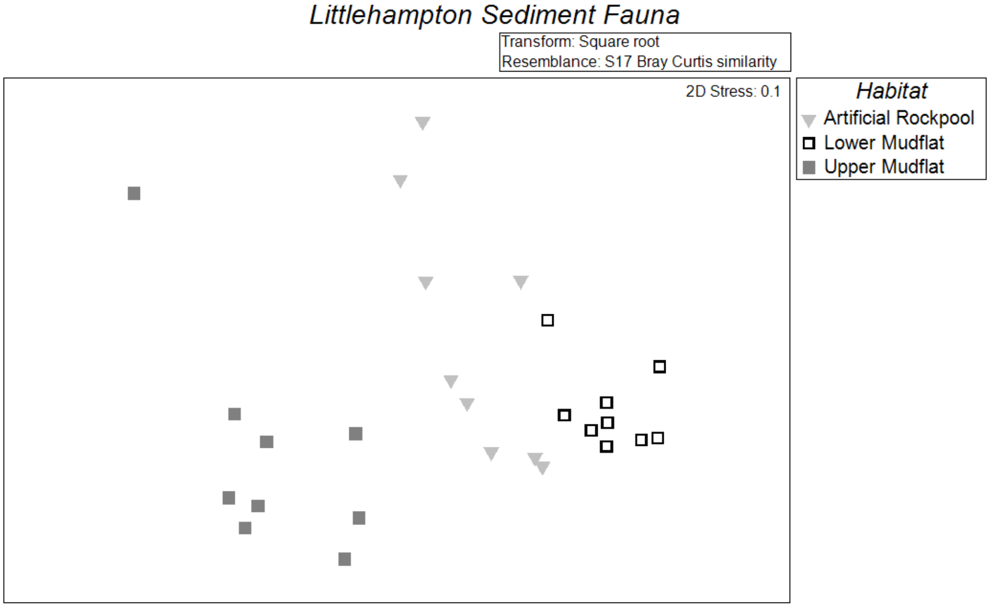

Figure 43. Multidimensional scaling (MDS) plots demonstrating the similarities of assemblages in the different soft-bottom habitats: artificial rockpool, lower mudflat, and upper mudflat (n=9).

A SIMPER analysis of the taxa which contributed most to the differences between habitat assemblages revealed that annelid worms consistently played key roles in all comparisons. The polychaete *S. shrubsolli* contributed the most with 30.8 % and 30.5 % of dissimilarity between artificial rockpools and lower mudflat, and between lower mudflat and upper mudflat respectively. Between the artificial rockpool and upper mudflat habitat, the ragworm *H. diversicolor* and the gastropod *P. ulvae* contributed 25.4 % and 22.4 % to dissimilarity respectively. All other taxa between habitats contributed less than 20 %.

7.5 Discussion

Although small-scale, to the best of the authors of this present work's knowledge, this is the first study to demonstrate that estuarine mud retained within artificial rockpools can provide adequate habitat for benthic fauna, comparable to adjacent natural soft-sediment mudflats. Many infaunal species can be found at depths exceeding the depth sampled in this study (5 cm) (Hines and Comtois 1985; Meadows et al. 1990; Cardoso et al. 2010), which was constrained by the maximum depth of the rockpools. Burrowing behaviour may also depend on the season and size of individuals, for example, larger ragworms may burrow to a depth more than 25 cm in winter, particularly in sandier sediments (Esselink & Zwarts 1989). Temporal variation in soft-sediment assemblages is well documented (Garcia-Arberas and Rallo 2002; Ysebaert and Herman 2002; Magni et al. 2006; Knott et al. 2018) therefore sampling over multiple depths, seasons and years would have been beneficial. However, this would cause significant disturbance to developing habitats within the artificial rockpools.

The limited volume of sediment retained in the artificial rockpools of this study may modify predator-prey or competitive relationships, as well as limiting fitness and growth (Strain et al. 2017). However, if scaled-up, enhancements specially designed to retain estuarine sediment may offer considerable potential to provide habitat and prevent biodiversity loss within heavily modified waterbodies. Although the soft-bottom infauna assemblages of the artificial rockpools differed to the local mudflats in species richness, abundance, diversity indices and assemblage structure, considering the patchy, variable nature of benthic assemblages, the rockpools nevertheless created a relatively analogous habitat to the top few centimetres of natural mudflat habitat. Considering the depth sampled, the faunal assemblages of the artificial rockpools were fairly typical of disturbed, organically enriched estuarine mudflat assemblages recorded in European estuaries and elsewhere (van der Linden et al. 2012). Although sediment characteristics (median grain size and organic matter content) were statistically significantly different between the rockpools and mudflats, the magnitude of this difference was likely to be relatively inconsequential. All habitats were silt-based (Wentworth 1922), and organic matter ranged across all habitats by ~1.5 %, remaining within organic matter values expected within enriched, estuarine mud (Hossain et al. 2014; Perkins et al. 2014; Coblentz et al. 2015).

The artificial rockpool assemblage structure and Shannon Weiner diversity index were more similar to that of the lower mudflat than the upper mudflat, despite the artificial rockpools being at similar shore heights to the area sampled at the upper mudflat. Additionally, the Shannon Weiner diversity index suggests that the artificial rockpool habitats were more diverse than the upper mudflats. Although their position in the tidal range was higher than the lower mudflats, the artificial rockpool mud drainage was restricted and so the mud appeared to remain more saturated at low tide. Reduced desiccation stress may enhance survival of intertidal organisms, which may be particularly pertinent with rising air temperatures (Intergovernmental Panel on Climate Change 2014; Maggi et al. 2016).

The assemblage structure within the artificial rockpools appeared to be more variable than the lower mudflat, as indicated by the PERMANOVA similarity values and MDS plots. This was likely due to the fact that they were discrete habitat units, as opposed to being a contiguous habitat like the mudflats, which may also augment the effects of shore height on what is able to colonise the artificial rockpool mud. Tidal height differences between the rockpools may also have contributed to this variability though was not tested due to low replication. The higher number of taxa that were present in the artificial rockpools may be because each artificial rockpool was colonised independently of each other, and by different species. Subsequently, this made the comparison of species diversity between the artificial rockpools and mudflats challenging as they operated at different spatial scales with each artificial rockpool representing a discrete 'island' and the mudflats representing a large, contiguous area (Rosenzweig 1995).

High abundances of small *H. diversicolor* individuals in the artificial rockpools suggested recent recruitment. The mechanism by which the artificial rockpool mud was colonised by infaunal species is unknown but may occur through adult organisms migrating from the mudflats, larval settlement, or through the incidental deposition of individuals that have been dislodged from bedload transport. Aberson et al. (2011) determined that *H. diversicolor* can disperse via swimming into raised sediment, though this tends to occur in ragworms <6 cm in length. It is possible that the steel sheet piling creates a favourable depositional environment within the in-pans, as greater sediment accretion was observed to have occurred at the piling toe within the in-pans. However, this is dependent on a multitude of interacting environmental conditions and will not occur everywhere all the time, as scour at the toe of piling structures is also well documented (Heery et al. 2017).

In a future where sea level rise meets increasingly developed coastlines, coastal squeeze limits the intertidal zone to a narrow vertical area on ACSs, leading to a net loss of intertidal soft-bottomed habitat (Doody 2004; Schleupner 2008; Pontee 2011). As evidenced in this study, it is possible to provide sediment-retentive features on vertical ACSs that offer intertidal habitat that is relatively analogous to estuarine mudflats, particularly where there is risk of soft-bottomed habitat loss. It was also noted that the exterior of the artificial rockpools were well colonised, with the brown alga *Fucus* *spiralis* and high abundances of barnacles, compared to comparatively sparse coverage of the same species on the sheet piling, which warrants further investigation. Increased abundance present on retrofitted eco-engineering interventions has been noted elsewhere on the south coast of the UK (Hall et al. 2019). Therefore, these types of interventions may also provide novel habitat on their exterior, maximising their benefits to enhancing faunal colonisation on ACSs. Concerns have been raised that ACSs have introduced non-local assemblages, such as native rocky shore species and invasives, to an area where they were previously uncommon (Bulleri and Chapman 2010; Airoldi et al. 2015; Perkins et al. 2015). Sediment-retentive interventions may promote and conserve the colonisation of local soft-bottomed species.

7.5.1 Further study and application

This study demonstrates proof of concept that sediment can be retained in vertically elevated eco-engineered features which can provide a relatively analogous mudflat habitat containing an assemblage of infauna that typically occur in temperate estuarine mudflats (van der Linden et al. 2012). Further study would benefit from replication on a larger scale with multiple sampling campaigns to consolidate the results, and to install deeper rockpools (e.g., 10-30 cm) that would enable infaunal colonisation and sampling to greater depths. The exterior/ underside of the sediment retaining feature may also be designed to provide habitat for species that prefer hard substrata, maximising the value of a sediment-retaining eco-engineering intervention (Hall et al. 2019). Additionally, there may be opportunity to trial the efficacy of other soft-bottomed habitats, such as elevated seagrass, saltmarsh, or reed beds by seeding retained mud with young plants, particularly in locations where these habitats are experiencing loss. Further study should also include close collaboration with coastal engineers and asset owners to determine both long-term and large-scale feasibility of adding these interventions to sheet piling, and how they may affect corrosion rates and loading stress. This collaboration is essential to the success of eco-engineering schemes and eventual incorporation of such interventions in planning and policy (Naylor et al. 2012; Perkins et al. 2015). At this pilot stage, it is unclear what additional benefits artificial mudflat habitats may provide; for example, food provision for birds and fish, or use for commercial hand-gathering of ragworms as bait (Scaps 2002).

The vertical extension of soft-bottomed habitats via eco-engineering may assist with meeting the enhancement requirements that are increasingly required of new construction. However, the incorporation of sediment-retaining interventions should not be used to justify new-build coastal structures when alternatives, such as soft-engineering, haven't already been considered, and further pilot testing is required (Firth et al. 2020). It should be noted that Littlehampton harbour, like many recreational marinas (Heery et al. 2017), is exceptionally sheltered and sediment retention is unlikely to occur on medium to high exposure shores, and therefore sediment retaining interventions will not be appropriate for every proposed site. Prior to adding such features on an existing coastal structure or integrating them into planning, it would be prudent to determine local sediment budgets and the depositional environment of the proposed site. For example, Waltham and Sheeves (2018) recorded mean yearly net sediment accumulation of between 17.5 mm yr^{-1} and 22.5 mm yr^{-1} in eco-engineered rockpools deployed intertidally on the coast of Townsville, Australia. However, in the present study, full siltation of the artificial rockpools was observed after just 1 year (60-70 mm). It is recommended that the local soft-sediment environment should be sampled to provide an indication of the infauna that could be expected to colonise the sediment-retaining features on the structure itself. Any intervention intended for sediment retention would not be appropriate on an ACS that is expected to facilitate soft-sediment erosion, as it is unlikely they would perform as expected.

7.6 Conclusion

The pilot installation of nine artificial rockpools on steel sheet piling have demonstrated they can enhance the structure by retaining soft sediment, that creates a habitat relatively analogous to an estuarine mudflat, albeit very small-scale. These results show that eco-engineering interventions can create both hard substrate and soft-sediment habitats on vertical ACSs, which provides an additional enhancement option for ACSs and the potential to develop design templates for new ACSs that might offer integral habitat-forming features. However, further pilot testing is required to design eco-engineering features specifically designed to retain soft sediments on hard infrastructure in heavily modified water bodies. Collaboration with coastal practitioners would be essential to maximise the efficacy of such schemes.

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7.8 Supplementary material

*Table S31. Tukey post hoc results with habitat as fixed factor for variables organic matter content and silt content. (*** = P<0.001, ** = P<0.01, * = P<0.05, NS = Not significant.)*

Groups	diff	lwr	upr	p adj
Water Content				
Rockpool – Lower Mudflat	7.577361	2.457237	12.697485	**
Upper Mudflat – Lower Mudflat	-2.531111	-7.498361	2.436139	NS
Upper Mudflat – Rockpool	-10.108472	-15.228596	-4.988348	***
Organic Matter Content				
Rockpool – Lower Mudflat	1.4420301	0.4453728	2.4386873	**
Upper Mudflat – Lower Mudflat	0.6208507	-0.3460488	1.5877503	NS
Upper Mudflat – Rockpool	-0.8211793	-1.8178366	0.1754779	NS
Median Grain Size				
Rockpool – Lower Mudflat	-7.651111	-8.340508	-6.961714	***
Upper Mudflat – Lower Mudflat	-3.288889	-3.978286	-2.599492	***
Upper Mudflat – Rockpool	4.362222	3.672825	5.051619	***

*Table S32. Tukey post hoc results with habitat as fixed factor for variables species richness and numeric abundance. (*** = P<0.001, ** = P<0.01, * = P<0.05, NS = Not significant.)*

Groups	diff	lwr	upr	p adj
Species Richness				
Rockpool – Lower Mudflat	-1.444444	-2.921425	0.0325362	NS
Upper Mudflat – Lower Mudflat	-3.222222	-4.699203	-1.7452416	***
Upper Mudflat – Rockpool	-1.777778	-3.254758	-0.3007971	*
Numeric Abundance				
Rockpool – Lower Mudflat	-3.774679	-6.752948	-0.7964108	*
Upper Mudflat – Lower Mudflat	-4.446980	-7.425249	-1.4687119	**
Upper Mudflat – Rockpool	-0.672301	-3.650569	2.3059674	NS

Chapter 8 – Discussion

This thesis has explored the relationship between concrete substrates and the marine life that colonises them, as well as increasing the understanding of how artificial rockpools may provide habitats at high tide, when retaining mud and at different tidal levels. This was achieved through extensive literature reviews supported by *in situ* experimental work, and through ecological monitoring of artificial rockpools using traditional quadrat surveys, underwater video, and sediment coring.

Following the review of bioprotective and biodeteriorative impacts of marine colonisation (Chapter 3), it is evident that facilitating and encouraging colonisation should continue, with colonising organisms and communities providing ecosystem services, including bioprotection. Coastal infrastructure that embraces bioreceptive features and finishes are more likely to host a more taxonomically and morphologically diverse community, which may in turn increase bioprotective properties further still (Chapter 2). If enhancements are integrated within a structure, such as a seawall, it is likely that more concrete will be used to ensure the minimum thickness requirement of concrete cover over steel reinforcement is met in concave or recessed areas of the seawall surface. The additional concrete cover on convex and protuberant seawall features may mitigate against the deleterious impacts of consequent colonisation by providing an extra physical barrier. However, increased use of concrete for ecological enhancements should come with the caveat that the carbon footprint of the development is consequently increased, and even with complete coverage of sessile communities and high biomass, it is unlikely to offset the additional carbon. Although many 'off the shelf' commercial products often use lower-carbon concretes, large flood defence schemes incorporating ecological enhancement through the use of formliners are unlikely to use the same lower-carbon concretes due to lack of durability, structural and strength testing, lack of awareness, reluctance to try new materials and higher costs.

The review of bioreceptivity (Chapter 2) and the experimental comparison of mortar suspended in the subtidal environment (Chapter 4) found that chemical bioreceptivity of concrete performs inconsistently and is an unreliable method by which to enhance bioreceptivity. Focus should remain on minimising the carbon footprint of concrete through the inclusion of recycled materials that do not deter colonisation, and by improving durability and extending service life to minimise repair and replacement and therefore disturbance to established biological communities.

This research has shown that retrofitted, bolt-on artificial rockpools can provide more than habitat for intertidal taxa at low tide only. They support the life histories of several mobile species at high tide (Chapter 5), with the adjacent seawall also playing a complementary role by providing ideal habitat for prey species (barnacles), which supports existing literature that recommends a mosaic of habitat features and substrate rugosities to cater to a diverse range of organisms' needs. In some sites (Littlehampton in Chapter 7 and Hamble Harbour in Chapter 6), basins or depressions intended to store water at low tide also store sediment, which can provide soft-sediment habitat for infauna. This presents an opportunity to artificially recreate soft-sediment habitats, instead of the sometimes-inappropriate intertidal rocky reef habitats and communities often used in intertidal eco-engineering when the structure footprint is replacing soft-sediment. However, there is no guarantee that soft sediment will be retained in artificial rockpool structures without prior assessment. The tidal elevation of artificial rockpools should also be carefully considered. This thesis has demonstrated that rockpools in the upper tidal zone can provide beneficial habitat refugia which will enhance a structure's climate resilience in the face of increasing sea level rise and air temperatures anticipated in the coming decades (Chapter 6).

This thesis has further demonstrated the ecological benefits of bolt-on artificial rockpools (Chapters 5, 6 and 7), which can be easily retrofitted to flat, vertical artificial structures, such as seawalls. An abundance of existing research has demonstrated the proof-of-concept of artificial rockpools as successful ecological enhancement, but retrofitted, bolt-on rockpools are not always the appropriate method of achieving water or sediment retention and rockpool habitat. To date, retrofit has been used extensively for existing coastal infrastructure, particularly in research and pilot trials, but is unlikely to be the best option for new build concrete coastal infrastructure. Artificial rockpools can be integrated into prefabricated or *in situ* seawalls using commercially available or bespoke form-liners which can be reused many times, allowing a large project to scale up integration of ecological enhancement efficiently and effectively (MacArthur et al. 2019). Cost considerations should factor in the additional concrete needed to achieve minimum cover over the steel reinforcement when using a form-liner.

In the UK, new developments are now required to achieve a 'biodiversity net gain' of 10% compared to the existing site (Environment Act 2021). At the time of writing, ecological enhancement was poorly provisioned in the government metric tool that calculates the percentage of net gain in proposed developments, leading to low scores. This could disincentivise developers from implementing ecological enhancement on coastal developments and instead seeking to offset their impact through terrestrial habitat creation, compounding intertidal habitat loss. Ranking enhanced coastal infrastructure higher within the metric would improve the overall score and therefore encourage uptake of ecological enhancement within coastal development. To promote integration of artificial rockpools into coastal development, the metric should be guided by the continually growing body of evidence that shows how significantly enhancement can increase biodiversity and the provision of ecosystem services. Further, it is essential that coastal practitioners and engineers engage early with appropriately qualified ecologists, so the integration of ecological enhancement is considered, bespoke, interdisciplinary and satisfies budgetary limitations and structural requirements. As noted by Naylor et al. (2012), inclusion of ecological enhancement on developments may improve planning application and funding outcomes. As public awareness of ecological and climate issues increases (Calculli et al. 2021), it will be in the interest of developers to maximise sustainability and ecological sensitivity to increase chances of social acceptance, particularly as coastal development can be met with significant resistance (Bush and Hoagland 2016; Hooper et al. 2020).

Aside from legislative drivers, non-mandatory drivers could include the impact ecological enhancement can have on people and place, by softening the harsh appearance of artificial structures with diverse, well-colonised communities and improving people's connection with nature, particularly on urbanised coastlines. Ecological enhancement should be considered enhancement for all species that use the coastal environment and thus facilitating the safe use of ecological enhancements by people should also be a conservation priority. 'Use', in this instance, does not necessarily mean anthropogenic disturbance of artificial habitats, but the ability to view them and understand their function and so appreciate and draw positive associations from them (Aguilera et al. 2023). Enhancement should therefore be incorporated in development in tandem with appropriate interpretation and early public engagement (Kosova et al. 2023), to maximise the social benefits the public may derive from enhancements (Strain et al. 2019) and establish social legitimacy (Kenny et al. 2023). Many coastal settlements experience deprivation (Agarwal and Brunt 2006; Agarwal et al. 2018) due to their isolated geography, but new developments have the potential to drive positive community changes (Green et al. 2020). If coastal practitioners are indeed willing to incorporate ecological enhancement, why stop there? As demonstrated by the work of Green et al. (2020), it is possible to holistically approach coastal development to incorporate cultural and social benefits and mental well-being (Scheres and Schuttrumpf 2019; de Bell et al. 2020). However, it is important to sensitively evaluate the will of local people, their cultural identity and their sense of place (Kibler et al. 2018) to avoid presumptuous design features that may negatively impact public perception.

8.1 Future work

It is evident that concrete chemical bioreceptivity can be inconsistent and spatio-temporally limited, and therefore further work should focus away from this field. If research in chemical bioreceptivity is pursued, the concrete should be *in situ* in a marine environment for at least one year to ensure the results more accurately reflect a broad range of stochastic and deterministic processes likely to be encountered during the lifetime of the structure. The relationship between substrate and epiliths is worthy of continued investigation, particularly incorporating experiments which examine climate-change scenarios and the impacts warming and CO_2 rich air and seawater have on marine concretes. Concrete is at significant risk of accelerated deterioration as a result of climate change (Stewart et al. 2011), and biological colonisation may provide a greater magnitude of bioprotection against these risks through cooling and the formation of a physical sessile barrier.

Sedimentation of water-retaining ecological enhancements has been reported elsewhere (Firth et al. 2016; Hall et al. 2018; Waltham and Sheaves 2018) as well as two of the studies included in this thesis (Chapters 6 and 7). These works offer an early picture of how retained sediment may provide habitat, but this would benefit from replication at other sites where enhancements have retained mud or sand to elucidate which environments this may occur in more frequently and which taxa are able to colonise sediment-retaining enhancements. This will enable more effective design of enhancements that are intended to retain sediment, which may mitigate against the soft-sediment habitat loss that so often occurs as a result of coastal development. Finally, monitoring of intertidal enhancements at high tide using video should be replicated at sites where communities are well established, to consolidate the findings recorded in this thesis (Chapter 5). To build on this, it would be beneficial to capture video during nighttime high tides, as many marine organisms behave differently during this time and therefore may interact with enhancements differently compared to the day. For similar reasons, video capture during the winter would also help build a picture as to how enhancements can provide habitat throughout the tidal cycle. Monitoring of colonisation from the point of installation should continue at a minimum on an annual basis for at least five years, and in some cases longer where sites are more exposed or prone to disturbance (once every five years thereafter). Climax communities take several years to form (Hawkins et al. 1983; Pinn et al. 2005; Coombes et al. 2011; Ferrario et al. 2016) and without capturing longer term data the capacity for understanding how ecological enhancements perform and be improved will be limited. Compulsory ecological monitoring for artificial coastal structures could be mandated for through the various licensing procedures necessary for coastal development, which could also stipulate the resolution of data as well as requiring it to be submitted to a governing body to ensure compliance and data sharing. Many artificial coastal structures require maintenance checks in which ecological monitoring could be incorporated, which could satisfy budgetary constraints.

Further investigation into the role of artificial rockpools, and ecological enhancements generally, as vectors for non-native invasive species is also needed. Ecological enhancements are considered to facilitate the colonisation of native species over non-native invasive species (Furchert 2019) but there is little evidence to date that this occurs. Although a contentious and complex issue, some non-native species may be considered beneficial, particularly where they form biogenic habitat and a bioprotective patina such as non-native oysters.

8.2 Conclusion

Intertidal habitat loss and replacement with coastal infrastructure will continue to occur, particularly in areas vulnerable to coastal erosion and sea level rise. This thesis demonstrates that it is possible to augment the bioreceptivity of concrete structures through varying surface rugosity and adding habitat features, such as artificial rockpools. Material choice should focus on cements and additives that are sustainable, promote longevity and climate change resilience. Through encouraging colonisation, coastal infrastructure will benefit from a diverse sessile community that may provide a bioprotective patina. Artificial rockpools should be installed at various tidal heights, including upper shore, to allow species to migrate vertically and find refugia where intertidal conditions are harshest. They may also retain sediment which can provide a sufficient habitat in its own right, in addition to providing habitat for mobile fauna at high tide. Integration of artificial rockpools into new build seawalls is possible through early design and consultation with ecologists, which negates the need for retrofitted bolt-on solutions. Ecological enhancement should be accompanied with considerations for social and public benefits wherever possible, and at minimum signage and interpretation, as well as ecological monitoring to determine enhancement success.

8.3 References

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