Geodiversity and biodiversity interactions: how natural rocky shore microhabitats can inform the ecological enhancement of engineered coastal structures

by

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Two published papers contain some of the research included in this thesis:

MacArthur, M., Naylor, L.A., Hansom, J.D., Burrows, M.T., Loke, L.H.L., Boyd, I., 2019. Maximising the ecological value of hard coastal structures using textured formliners. *Ecol. Eng*. X 100002. https://doi.org/10.1016/j.ecoena.2019.100002

Naylor, L.A., MacArthur, M., Hampshire, S., Bostock, K., Coombes, M.A., Hansom, J.D., Byrne, R., Folland, T., 2017. Rock armour for birds and their prey: Ecological enhancement of coastal engineering. *Proc. Inst. Civ. Eng. Marit. Eng.* 170, 67–82. https://doi.org/10.1680/jmaen.2016.28

Abstract

The primary aims of this thesis were two-fold. First, this thesis examined the interplay between geodiversity and biodiversity on natural rocky shores from regional – site – microhabitat scales exploring how rock material and rock mass properties influence geomorphological and biogeomorphological processes that shape microhabitats, which in turn exerts an influence over biota. A key finding from this part of the thesis was that lithology is not an important determinant of species richness and abundance at the national or regional scale, being more a modifier of patterns than a driver. At the site scale, lithology exerts more of an influence, with complex lithologies (such as limestone) and associated microhabitats more likely to have higher species richness and abundance than adjacent areas of the shore platform that lack this geomorphic complexity.

All the work in this thesis was conducted in the mid-upper intertidal zone, between Mean Tide Level (MTL) and Mean High Water Springs (MHWS), as this zone has high exposure during low tide and most coastal defences are built at this level, meaning interventions at this tidal height hold the most biodiversity potential. Natural shore surveys were also conducted at this height to directly inform engineering design and to contribute to biogeomorphological theory as more species were found to congregate in microhabitats for refuge in the mid-upper intertidal zone.

Results of surveying the mid-upper intertidal zone of several rocky shores across the UK highlight that it is critical to account for both lithology and the presence of geomorphic features (microhabitats) to better understand the distribution of species and their habitat requirements. The location of species within quadrats on each shore was recorded relative to their position on either the shore platform surface or within microhabitats, such as pools, cracks, pits and ledges. Results showed that microhabitat type is a key driver of species distribution within shores, while lithology modifies patterns between shores (10s of km scale). Although the most suitable habitat varied with location, deep pools (2.8-24 cm deep) were significant in increasing species richness and abundance where they were present while crevices and ledges facilitated significantly greater mobile species abundance, particularly compared to the adjacent shore platform. These findings improve the understanding on the interrelationships between geodiversity and biodiversity and highlight the need for ecological and

biogeomorphological surveys to incorporate species distributions within geomorphic features and processes in greater depth.

The second aim drew from this new understanding of the rock and microhabitat preferences of intertidal species on natural rocky shores to evaluate the effectiveness and further the evidence base for ecological enhancement, i.e. improving the quantity and quality of available habitat for species on artificial structures, on new and existing artificial coastal defences.

With the construction and expansion of coastal defences in the intertidal zone globally, ecologically enhanced designs are needed to mitigate some of the impacts of construction on the intertidal zone, such as disturbance from the addition of artificial substrate and habitat loss. To do this, an ecological enhancement trial using 160 artificial concrete tiles of 8 different designs and 24 cleared natural surfaces was conducted at three UK sites over an 18-month period, representing the largest (to date) UK enhancement trial of this kind. Key findings from this trial showed that intermediate complexity in the form of mm-scale grooves was statistically significant in increasing the abundance of early-colonising species (i.e. barnacles) from 2 months onwards compared to plain-cast control tiles and designs of higher complexity. Additionally, the design with the highest level of habitat complexity (up to 30 mm deep pits) significantly increased humidity and reduced temperature compared to lower complexity designs. Species richness and abundance was greatest in the microhabitats of the most complex design during monitoring from 2-18 months after installation. These designs highlight the value of ecological enhancement from the mm-cm scale in providing habitat that would otherwise be absent on plain-cast artificial coastal structures, such as seawalls.

A secondary ecological enhancement trial was conducted on passively enhanced (passive positioning and optimised material choice for ecology) rock armour boulders at a live coastal defence scheme in Hartlepool. From laboratory testing, Portland limestone and Carboniferous limestones (Hartlepool and Welsh) were optimal boulder material choices for rock armour revetments, combining ecological and engineering suitability. Adding in field survey results showed the importance of considering ecological suitability from the mm-dm (decimetre) scale and including rock material and rock mass properties in making engineering recommendations. Ledges at the dm-scale in field surveys on the

revetment at Hartlepool were the optimal geomorphic feature in significantly increasing limpet abundance on the boulder surface. To conclude, the findings from natural shores were utilised to suggest improvements to future ecological and biogeomorphological survey techniques on natural and artificial shores. These in-depth surveys were coupled with findings from ecological enhancement trials to provide detailed recommendations on the design of future enhancements on artificial shores, with specific design parameters delineated.

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Chapter 1. Introduction and Aims

The overarching aim of this thesis is to investigate how the lithology and microhabitat preferences of intertidal species on natural rocky shores can help inform ways in which new and existing artificial coastal protection structures may provide ecological enhancement. This work further aims to highlight the importance of including rock material and rock mass properties in ecological studies and identify the need for more detailed habitat sampling on natural rocky shores. This research will assess the effectiveness of methods of enhancement that increase the quantity and quality of available habitat on artificial structures and enhance both ecological and biogeomorphological understanding of biodiversity-geodiversity interactions on rocky shores. To do this the nature of rocky intertidal shores and their intimate interrelationship with ecology need to be outlined in advance of introducing the aims, methods and delivery of ecological enhancement to structures.

1.1 Rocky intertidal shores

Rocky shores constitute approximately 42% of the British coastline (Jackson and McIlvenny, 2011) and an estimated 80% of the global coast (Emery and Kuhn, 1982). These hard substrate shores offer a range of important ecosystem services including: functioning as natural sea defences; larval supply; biomass production; and as an important feeding habitat for birds and fish species due to their high productivity and structural complexity (Branch et al., 2008; Fletcher et al., 2012; Seitz et al., 2014). These shores are highly varied in their wave exposure, lithology and gradient resulting in a rich diversity of landform features including rock pools, stacks and arches (Kennedy et al., 2014). This also results in variable levels of topographic and habitat complexity that serves to support diverse species assemblages (Jackson and McIlvenny, 2011).

Topographic complexity is the arrangement and diversity of structural elements over a surface (Zawada et al., 2010) and is used by ecologists to define geomorphic complexity. This geomorphic complexity arises from a complex interplay between rock material, rock mass and environmental processes acting to shape landforms such as shore platforms, the erosional features formed by the retreat of cliffs over time resulting from wave action. This complexity includes sub-landform features such as rock pools (gathering of water in surface depressions), pits (hollows in the platform surface) and crevices (narrow openings in the shore platform surface). Geomorphic complexity (synonymous with topographic complexity) and habitat complexity, the number and diversity of habitat types, are components of geodiversity. Habitat complexity is analogous to beta diversity, with habitat complexity influencing species distribution and diversity between different microhabitats (Henderson et al., 2017).

Geodiversity is the physical equivalent of biodiversity and is defined as the variation in materials, landforms and processes on Earth (Gordon et al., 2012; Gray, 2013). This incorporates geological and geomorphological diversity (i.e. landforms and their variation in space and time, Corenblit et al., 2011; MacArthur et al., 2019; Scheffers et al., 2012). Geodiversity is widely recognised as providing the physical underpinning that supports most ecosystems and species, with complexity in the physical environment closely linked with biotic complexity (Hjort et al., 2015; Parks and Mulligan, 2010). In the global drive to conserve biodiversity, it is important to note that geodiversity is an integral part of nature that supports and delivers fundamental ecosystem services and is crucial for sustaining living species and their associated habitats (Gordon et al., 2012; Hjort et al., 2015).

Geological properties of rocks, hereafter geology, is a key part of geodiversity and includes rock material properties (the properties of the rock itself such as lithology, porosity, colour) and rock mass properties (the macroscopic features of the rock mass, such as crevices, cracks, pits and pools , United States Department of Agriculture, (2012)) which influence processes on rock coasts (Naylor et al., 2012b). Variability in geology can result in variation in coastal morphology and resulting landforms, as the geology and its associated factors (e.g. lithology and jointing) can determine the speed at which landforms erode (Kennedy et al., 2014). One example of this is the formation of shore platforms by cliff recession as a result of wave action, with the rate of recession substantially greater on soft rock cliffs (Sunamura, 2015). Although rock hardness generally increases the erosional resistance of the rock, the presence of jointing and fractures introduces zones of weakness which increase erodibility and can exert a greater influence than lithology (Kennedy et al., 2014). They also provide finer scale geological controls that influence the development of sub-landform features like pools (Naylor, 2001).

Waves are significant forces in the development of shore platforms through hydraulic action, with erosion occurring where the force of waves exceeds the resistance of the rock (Sunamura, 2015). A suite of geomorphic processes (physical, biological and chemical weathering) contribute to the reduction in the resistance of rocks and influence rock mass properties by creating zones of weakness on the shore that increase the erodibility of rocks by wave-induced action (Naylor et al., 2012b); creating topographic complexity.

In briefly examining the influence of each of these processes, physical weathering can be broken down into freeze-thaw, wetting and drying and salt crystallisation. Freeze-thaw involves the expansion of water within joints and pore spaces in the rock as it freezes, which would have been a key weathering force during past cold periods in Britain but would exert a lesser influence in the current temperate climate (Coombes, 2014). Under experimental conditions, zones of weakness in the rock coincided with concentrated deterioration of the rock, this is particularly evident in stronger lithologies such as crystalline limestones (Nicholson and Nicholson, 2000). Wetting by the tide and rain and subsequent drying through insolation and exposure to air results in the expansion and contraction of minerals (Coombes, 2014). In temperate conditions, metamorphic and igneous lithologies experience limited breakdown by wetting and drying (Kanyaya and Trenhaile, 2005) but shales, mudstones and argillaceous lithologies are more susceptible to this form of weathering (Coombes, 2014). Salt weathering occurs where saline sea water enters rock discontinuities (planes of weakness), crystallises and then expands with heat. This can weaken or downwear sections of the rock surface and produces microcracking, deepens cracks and disintegrates the surface (Doehne, 2002). These weathering processes contribute to the widening and weakening of discontinuities in advance of wave action.

Chemical weathering occurs where minerals in the rock react with air or water and are dissolved or transformed via hydrolysis and hydration, oxidation and carbonation (Coombes, 2014). This weathering can increase the number and volume of pore space and decay discontinuities, with some minerals, such as feldspar, more susceptible to chemical decay than others, such as quartz and muscovite (Mottershead, 2000). Dissolution is particularly important in carbonated lithologies (Moura et al., 2006), which can increase susceptibility of the rock to additional weathering processes.

Biological weathering involves the weakening of the rock by flora and fauna that have important biogeomorphological roles (two way interaction between biology and the physical environment) (Moura et al., 2012; Naylor et al., 2012b; Pinn et al., 2008; Spencer and Viles, 2002). Bioerosion occurs at various scales, from microorganisms breaking down the outer layer of the rock (Coombes et al., 2011), boring species at the mm scale including *Boccardia* sp. being found in joints and potentially facilitating joint widening and weakening (Naylor et al., 2012b). Piddock bivalves also facilitate rock weakening by boring into soft rock environments (at the cm – dm (decimetre) scale) using their shells and creating holes in the rock (Pinn et al., 2008). Grazing species, such as limpets, further facilitate biological weathering by scraping the outer layer of the rock (Coombes, 2014; Naylor et al., 2012b; Scheffers et al., 2012).

The extent to which the platform is influenced by these processes and the associated rock mass properties, including microhabitat density (jointing and pool formation), and rock material properties including lithology and the strength of the rock, is highly geologically contingent (Naylor and Stephenson, 2010). Along these planes of weakness where marine erosion expands joints and cracks, rock mass may become detached from the platform surface, resulting in the formation of rock pools (Noormets et al., 2002). Weaker surfaces or less resistant rock will erode more rapidly than outcrops of more resistant rock on the platform surface, resulting in pooling of water in these depressions (Denny and Gaines, 2007). The presence of bioerosive species within these joints, particularly on softer lithologies such as limestones, can aid in the removal of rock mass to contribute to the creation of rock pools (Naylor et al., 2012b). Biogeomorphological interactions between softer, more friable lithologies and their residing species typically generates increased topographic complexity, as seen with Piddock burrows (Coombes, 2014; Pinn et al., 2008).

Each of these processes has varying contributions to shore platform generation and subsequent erosion, with the primary contributor (physical, chemical or biological weathering) varying by site due to the diversity of contexts globally (geological, geomorphological, climatic) (Moura et al., 2006; Trenhaile, 2018). Physical, chemical and biological weathering contribute to the generation of topographic complexity through the weakening and enlargement of planes of weakness (such as joints or cracks), which results in the creation of microhabitats, including crevices and pools, on rocky shores that generate a positive ecological response.

Rocky intertidal species are highly adaptable to extreme conditions. They must survive on hard substrates which limit the burrowing ability of species (other than the few known rock borers, e.g. Naylor et al., 2012b; Pinn et al., 2008; Trudgill, 1988a), and they are exposed to thermal and desiccation stress when exposed at low tide (Branch et al., 2008). In addition, the risk of physical stress, such as wave action (McQuaid and Branch, 1985), or biological stress, such as predation (Fairweather, 1988a; Hunt and Scheibling, 1998) is particularly prominent in the mid-upper intertidal zone, so the physical environment of the substratum is crucial for sustaining species and providing the habitats they depend on (Hjort et al., 2015). The intertidal zones of shore platforms are also subject to abrasion where wave energy is greatest, with shore geometry influencing wave dissipation (Blanco-Chao et al., 2006) and efficiency of abrasion varying with lithology, mineral composition, previous weathering and bedding planes (Feal-Pérez and Blanco-Chao, 2013). Smoothed abrasion zones on shore platforms have a characteristic absence of developed ecological communities, although gastropods and limpets or early colonising green algae, such as *Ulva* spp., may be present (Blanco-Chao et al., 2006).

Anticipated changes in climate will have a notable effect on intertidal zones, with future sea level rise expected to significantly reduce rocky shore intertidal habitat and alter community abundance, distribution and composition (Jackson and McIlvenny, 2011; Kaplanis et al., 2019). It is therefore important to understand the interrelationship between the physical environment of rocky shores and the species that depend on it, as some lithologies may be more biologically suitable than others. This has been observed on the Plymouth breakwater, where eroded limestone blocks formed diverse rock pools over time, whereas adjacent granite blocks had limited colonisation (Jackson, 2015; Moschella et al., 2005). Limestone and other carbonate lithologies weather faster than harder igneous lithologies, such as granite. This results in carbonate lithologies having greater potential to increase their complexity at the micro-scale (µm-mm) through bioerosion (Moschella et al., 2005; Naylor, 2005), which is the direct or indirect erosion of substrate by grazing or boring organisms (Trudgill, 1988b), and at the mm-dm scale as crevices and pools can form over time (Moschella et al., 2005). Limestone also has more ecologically suitable properties than granite, with lower sub-surface temperatures and greater water absorption capacity than granite, with limestone remaining wetter over time, which may reduce desiccation and thermal stress (Coombes and Naylor, 2012). The extent of the influence of these interactions depends on many factors including climate, shore position, substratum type and the effects will vary by individual species (Coombes et al., 2011; Coombes and Naylor, 2012).

Focusing on the substratum of intertidal rocky shores, the influence of geodiversity on biodiversity is twofold: through rock material properties and rock mass properties. Geomorphologically generated microhabitats, such as crevices and pits, are of particular importance in providing refuge for organisms against biotic and abiotic stressors during low tide and have been shown to reduce species mortality (Walters and Wethey, 1996) and provide sheltered, less physically stressful (i.e. wetter and lower temperature) microclimates than the exposed platform surface (Aguilera and Navarrete, 2007; Harper and Williams, 2001; Kostylev et al., 2005; Lively, 1986; Menge et al., 1983). This results in species distributions varying with the presence of microhabitats, with increased biodiversity and abundance typically found in microhabitats compared to adjacent areas without them (Firth et al., 2014a; Firth et al., 2014b; Hall et al., 2018; Liversage et al., 2017; Strain et al., 2017b).

1.2 Systematic Review

The following review was conducted to frame blue skies research questions and identify gaps in ecological literature surrounding surveys of microhabitats. A secondary aim of this review was to examine how published ecological studies deal with lithology, its associated characteristics and complexities. To determine how ecological studies incorporate microhabitat features, a systematic review was undertaken in April 2019 using Google Scholar and Web of Science. The ecological literature was searched for field studies in the intertidal zone and the search terms included ("rock coast" OR "rocky shore" OR "intertidal") AND ("habitat complexity") AND ("Rocky shore ecology" OR

"Intertidal ecology") AND ("habitat" OR "microhabitat*" OR "crevice*" OR "pool" OR "pits*" OR "substrate") AND ("survey" OR "quadrat"). The addition of quadrats and surveys was an exclusion criterion to examine sampling methods. The analysis included papers from the years 1990-2019 to focus on relatively recent literature. Web of Science returned one result (Wilding et al., 2010) that was excluded as it was included in the Google Scholar search. Of the 196 studies found through the literature search, a total of 63 studies were analysed following exclusion of studies for their unsuitable location, irrelevance or study type being inappropriate (Figure 1-1). These studies are listed in Table A 1-1.

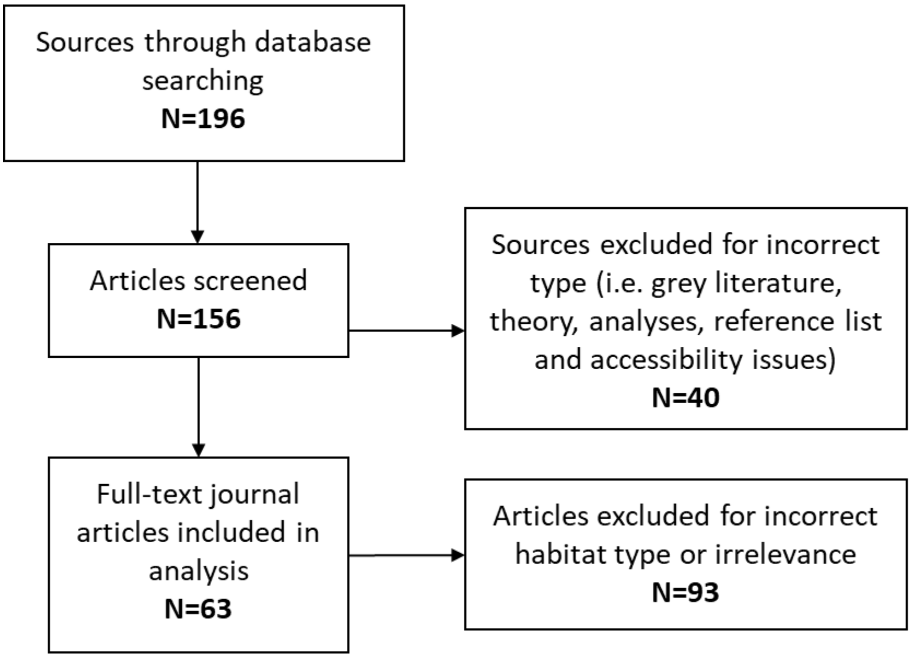

Figure 1-1. Process of exclusion for systematic literature review on sources from 1990-2019 in Google Scholar.

Of the 63 studies examined, 76% were conducted at multiple sites and 95% were on natural shores or conducted across natural and artificial shores (13%). Although the primary focus was examining whether the studies noted or discussed rock mass features, each study was additionally searched for the inclusion of several rock material properties (porosity, albedo, lithology) to see whether these properties were reported in studies as background information. Of all the studies, only one (Peglow, 2013) examined differences in lithology on intertidal community structure and highlighted that differences in substratum may drive rocky intertidal community structure. Lithology was greatly overlooked but 34% of studies mentioned the substratum type as either cobble, boulder etc. without providing much further detail on the geology or geomorphology, highlighting that there is a need to better consider lithology in ecological studies.

Other rock material properties including porosity and rock chemistry were overlooked by all examined studies and albedo, i.e. the light reflectance value of rock colour, or rock colour only considered in two studies (Heady, 2013; McAfee et al., 2017), both finding that colour can influence the settlement or distribution of species. Heady (2013) noted that the reflectance of rock colour accounted for 17% of the variability in the upper limit of high and mid intertidal zone species. McAfee et al. (2017) noted a higher recruitment of species to white substrate than black, with a thermal effect driving these differences. Accounting for the influence of colour is important in the face of a warming climate as darker lithologies will retain heat and influence the body temperature of species (Heady, 2013), with species on darker lithologies more susceptible to the effects of warming than species on lighter substrata (McAfee et al., 2017). This will ultimately impact the vertical distribution and survivability of intertidal communities in future.

In examining the role of microhabitats (approximately 1-100s cm after Evans et al., 2015), biogenic habitat (i.e. habitats created by fauna or flora) is widely acknowledged as being important. 27 studies mention or describe the role of biogenic habitat in providing refuge or increasing habitat complexity, particularly referring to macroalgal species (Ape et al., 2018; Kraufvelin and Salovius, 2004; Smith et al., 2014; Thrush et al., 2011). Algal turfs offer a more complex habitat, increasing the structural complexity of the shore and providing protection from predation and waves, offering suitable feeding surfaces for small macrofauna (Brown and Taylor, 1999; Buschmann, 1990; Kraufvelin and Salovius, 2004; Martinez, 2011) and maintaining moisture during low tide (Smith et al., 2014) which would reduce desiccation stress. This can result in increased macrofaunal diversity and abundance where there is a large occurrence of macroalgae (Kraufvelin and Salovius, 2004; Martinez, 2011). Mussels can also provide biogenic habitat as their complex shell structures increase shading, reduce rock surface temperature and trap moisture (Lathlean and Minchinton, 2012 (not in review); McAfee et al., 2017). Barnacle mosaics were also noted as providing important refuge habitat for small macrofauna, facilitating settlement and supporting a more diverse assemblage than bare rock (Hull, 1999; Martinez, 2011).

Biogenic habitat (e.g. algal turfs) received more attention than other intertidal habitat structures, with physical microhabitat provision largely overlooked as only 19 of the 63 studies looked at microhabitats. Two of these studies did not examine the biota residing in microhabitats and instead focused on measuring the abundance of habitat components like crevices or assessing methods for accessing recesses to quantify habitat complexity (Meager et al., 2011; Wilding et al., 2010). A few studies examined pits/holes (including Faria and Almada, 2001; Loke, 2015; MacArthur et al., 2019; Martins et al., 2010) or crevices (Aguilera et al., 2014; Faria and Almada, 2001; Glassey, 2002; Healy, 1996; Lee et al., 2009; Wilding et al., 2010), although a greater proportion acknowledged that these features provide important refuge for intertidal species (n=8 crevice refuge, n=5 pit/hole refuge). Pools were the most commonly sampled feature (n=15) and were found to support greater species richness than adjacent substrata (Firth et al., 2013) and increasing the upper vertical limits of some species when added to artificial substrata (Ostalé-Valriberas et al., 2018). A total of 10 studies noted that pools could provide a refuge function from physical stress but these habitats can provide their own physical stress due to fluctuations in temperature, pH and salinity amongst other factors (Firth et al., 2013).

Species richness was found to be variable at the km-scale, whereas abundance has been shown to be controlled by small-scale (millimetres (mm) to decimetres (dm)) substrate heterogeneity such as differences in microhabitat (Hull, 1999). There is a high amount of small scale variability on intertidal shores as higher level trophic species have a preference for particular microhabitats that can function as refuge (Burrows et al., 2009). The presence of microhabitats such as cracks, crevices, pools and overhangs can provide shade and hold water, with organisms found to concentrate around these features to reduce environmental stress (Peglow, 2013) and some species restricted to these microhabitats for survival (Anu et al., 2017). Yet, these physical features are typically under sampled in ecological literature but are important in buffering harsh environmental conditions and enhancing settlement of species, with a positive density dependency found between grazing species and microhabitat availability (Aguilera et al., 2014; Martins et al., 2010).

From the above review, it emerges that the effect of microhabitats on species distribution (i.e. offering increased refuge for species against desiccation stress, wave exposure and predation) are under sampled in ecological surveys and their value is under reported in ecological literature. To better understand the relationship between geodiversity and biodiversity on intertidal rocky shores, and the biogeomorphological interactions that occur between them, the interrelationship between microhabitats and rock mass/material properties need to be understood in depth.

1.3 Lithology, surface morphology, structural features and ecology

A number of factors at the global and regional scale will influence biogeographic differences in community composition and distribution including temperature, wave action, tidal range and salinity (Ramos et al., 2016). Where these factors are relatively uniform at the local scale, geology (lithology and structure), surface geomorphology and context may exert more of an influence on species distributions. Lithology can contribute to variation in the distribution and interactions of species at small and moderate scales as aspect and local topography can be highly variable at the cm to km scale (Schoch and Dethier, 1996). Geomorphology overlaps to some extent with surface geology in that the rock surface and topography is controlled by the bedding and jointing inherited from the rock itself as well as the geomorphological processes that have reworked and modified the surface over the period since the rock became exposed.

Individual lithologies vary in their material properties including colour, porosity, roughness and hardness. For example, rock colour influences the absorption of heat which then influences settlement and recruitment of organisms, as lighter coloured lithologies reduce thermal stress experienced by species (McAfee et al., 2017; Raimondi, 1988) since their higher albedos result in lower surface and sub-surface temperatures (McGreevy, 1985). Variability in heat retention and porosity between different substrate types influences the survivability, cover and composition of algal communities (Green et al., 2012; McGuinness and Underwood, 1986). Lithologies with more pore space stay wetter for longer, which can create a moister habitat and aid in survival during low tide (Coombes and Naylor, 2012). These more porous lithologies tend to be softer and more chemically favourable, such as limestones, and may also offer greater ecological engineering potential over time (Naylor et al., 2012b).

Surface roughness also exerts an influence on the settlement of early colonising species, i.e. barnacles, and the build-up of marine biofilm, with both aiding further community development (Cacabelos et al., 2016; Coombes et al., 2015). For example, boring species (e.g. cyanobacteria) on softer calcareous lithologies can increase the complexity of substrate at the micro-scale (< 1 mm), which can increase porosity and alter microclimate and subsequently rock wetting and drying: if borers are in great enough numbers they can weaken the rock (Coombes, 2011; Naylor et al., 2012b). Such micro-scale biogeomorphic interactions are effectively a form of ecosystem engineering that enhances habitat for macro-scale species by increasing surface roughness (Naylor et al., 2012b).

The influence of lithology on recruitment is species specific and can influence the geographical limits of species (Green et al., 2012; Herbert and Hawkins, 2006) and is more likely to influence community composition at the shore scale. Within individual shore platforms, the geomorphology of the coast, and the landform features such as crevices and pools that result from geological-geomorphological-ecological interactions will directly influence physical habitat complexity. This is likely to exert a greater role on species distributions and abundance, as was found from the above systematic review. This review also noted the biogenic habitat provided by algal turfs, barnacles and mussels can also provide a moister microclimate, reduced thermal variation during low tide and refuge from harsh intertidal conditions. Where there is a high cover of macroalgae in the lower shore, the refuge function of physical microhabitats for survival during low tide (i.e. moister microclimate and shelter) would likely be reduced.

All the work in this thesis was conducted in the mid-upper intertidal zone, between Mean Tide Level (MTL) and Mean High Water Springs (MHWS), as most coastal defences are being built at this level and as a result support lower biodiversity than structures lower in the tidal zone (Burcharth et al., 2007; Firth et al., 2013). Additionally, this area is exposed for lengthy periods during low tide and microhabitat availability will have a greater influence on species survival than at wetter lower tidal levels. Rocky shores are heterogeneous geomorphic systems (Kennedy et al., 2014) which influence the diverse range of habitat types available and varying degrees of physical habitat complexity found on a given rocky shore. This physical habitat complexity, along with biogenic habitat complexity and ecosystem engineering are thus fundamentally important in affecting community diversity, composition and the interactions between species (Firth et al., 2014b; Kostylev et al., 2005). However, not all habitat types will be equally beneficial to ecology during low tide and the type and quality of habitat available will influence biodiversity on the shore (Firth et al., 2014b). As both the physical habitat and ecosystem engineering potential on rocky shores has strong geological and geomorphological controls, it is important to draw in expertise from geology and geomorphology to better establish the complex interactions between ecology and habitat complexity.

The diversity of microhabitats on rocky shores, such as pits, pools, overhangs and ledges, are important geomorphological features on natural shores, offering a range of habitats for different species (Bugnot et al., 2018; Jackson, 2010; Raffaelli and Hughes, 1978; Rickards and Boulding, 2015; Underwood and Jernakoff, 1984). Pools offer constant submergence and reduced desiccation risk for marine organisms during low tide (Martins et al., 2007) and typically have a greater diversity and abundance of species than surrounding non-pool areas (Bugnot et al., 2018; Firth et al., 2014a). They offer refuge and important nursery and feeding habitats for intertidal species and extend the vertical distribution of some species (Firth et al., 2014a; Underwood, 1981). Nevertheless, pools can also be stressful environments during low-tide periods, as physio-chemical conditions, including pH, temperature and salinity, can vary greatly over short time periods (Firth et al., 2013; Martins et al., 2007) with shallow pools of limited volume offering little buffering against environmental conditions (Firth et al., 2013; Firth et al., 2014a). Pool depth also influences community composition, diversity and species survival (Martins et al., 2007; Moschella et al., 2005) with deep pools offering more stable environmental conditions, particularly in the upper shore (Martins et al., 2007).

Crevices, pits and holes offer refuge from desiccation stress, predation and scouring and aid in community development, with greater biomass, diversity and richness found in microhabitats than adjacent exposed areas (Archambault and Bourget, 1996; Cartwright and Williams, 2012; Prendergast et al., 2009; Rickards and Boulding, 2015). Crevices can reduce desiccation stress as they offer a cooler and more humid microclimate than external environmental conditions (Jackson, 2010). The width and orientation of these features can alter their microclimate, with a gradient of environmental conditions from the outside in (Healy, 1996). The opening angle and direction of crevices can also modify local hydrodynamic regimes by reducing wave exposure (Archambault and Bourget, 1996; Blanchard and Bourget, 1999). As a result of their unique microclimate and shelter provision, increased abundances of certain species and greater faunal diversity have been found within these topographic features compared to adjacent areas of lower complexity (Aguilera and Navarrete, 2007; Bulleri, 2005; Healy, 1996; Scrosati and Heaven, 2008). The impact of these effects on different species will vary with height on the shore, tidal conditions and season (Jackson, 2010).

1.4 Why is it important to examine the relationship between biodiversity and geodiversity?

The positive interrelationships between topographic complexity (one aspect of geodiversity) and species diversity and richness in many ecological communities has a substantial volume of theoretical support (Johnson et al., 2003; MacArthur and MacArthur, 1961; Menge et al., 1983; Pinn et al., 2008). Topographic complexity influences community structure and numerous ecological processes, such as providing refuge from biological and physical disturbance (Le Hir and Hily, 2005; Lee and Li, 2013; Menge et al., 1985) and creating variations in microclimate (Kostylev et al., 2005). Microclimatic variability is the variation in temperature and humidity between the platform surface and geomorphologically formed microhabitats. The refuge function provided by microhabitats reduces the influence of wave action and temperature whilst increasing relative humidity (Kostylev et al., 2005). This in turn alters the immersion period of species and levels of desiccation stress (Guichard et al., 2001). Softer lithologies, such as limestone, promote greater biogeomorphic ecosystem engineering, whereby micro-scale bioerosion generates increased surface roughness, improving the absorption of the material and allowing the surface to stay wetter for longer, facilitating a less stressful environment for macrofauna (Coombes and Naylor, 2012).

In order to protect and manage coastal biodiversity, it is fundamental to understand the geological, geomorphological and biogeomorphological factors influencing community composition. Under existing climate change scenarios, some intertidal habitat will be lost as a combination of sea levels rising, increased storminess (and potentially increased erosion) and the construction of coastal defences contributes to coastal squeeze (Jackson and McIlvenny, 2011). Rising sea levels will have a greater impact on natural rocky shores where shore platforms are backed by artificial coastal defences as there will be habitat loss in lower tidal zones. This in turn would have adverse impacts on wave, tidal and erosion regimes at the coast and subsequent shifts in the biogeographical ranges of species (Scottish Natural Heritage, 2015).

1.5 Ecological enhancement

Globally, natural intertidal habitat is being altered by the various types of coastal infrastructure (e.g. seawalls, groynes, rock armour and breakwaters) that are proliferating within the intertidal zone (Airoldi et al., 2005; Bulleri and Chapman, 2010; Chapman and Underwood, 2011; Moschella et al., 2005). This continued expansion and development of hard coastal infrastructure has associated ecological and geomorphological impacts, such as the reduction of available habitat due to the narrowing (or squeeze) of the intertidal and shallow subtidal zones (Jackson and McIlvenny, 2011).

The principles of ecological engineering are being increasingly incorporated into studies looking to enhance the ecological and biological value of coastal infrastructure. Ecological engineering, hereafter ecological enhancement, is a hybridisation of artificial structures, as components of natural ecosystems are incorporated into their construction and design. It is based on the need for engineered designs that "provide for human welfare while at the same time protecting the natural environment from which goods and services are drawn" (Bergen et al., 2001, p.201). In other words, the design methodologies of artificial structures need to meet the economic and social needs of society whilst enhancing the ecosystems they impact in order to improve the quality or quantity of available habitat (Firth et al., 2014a; Hall et al., 2018).

Species assemblages on artificial and natural substrates are notably different and with lower diversity and abundance of species on artificial coastal infrastructure compared to natural habitats, they are poor ecological surrogates for natural shores (Bulleri and Chapman, 2010; Chapman and Blockley, 2009; Coombes et al., 2015; Firth et al., 2014b;

Lai et al., 2018). The key reason for this variation is the lack of topographic complexity of the substrate leading to a lack of important microhabitats. This results in limited habitat complexity on artificial coastal structures compared to natural rocky shores, such as fine-scale (mm-cm) texture and microhabitats at the cm-dm scale. Thus, artificial structures are typically poor surrogates of the geomorphic heterogeneity often present on natural shores.

Current methods of enhancement have largely focused on manipulating texture or design of engineered materials or structures to attract an increased diversity and abundance of species from natural assemblages (e.g. Firth et al., 2016a; Naylor et al., 2011; Perkol-Finkel and Sella, 2014; Strain et al., 2017b). This is being considered at multiple scales and for multiple infrastructure and landform types in the marine and coastal environment. This includes marine installations such as eco-designing offshore windfarm foundations to mitigate any environmental degradation that might occur from their deployment (Lacroix and Pioch, 2011; Langhamer, 2012) or the use of designed concrete (Reefballs™) in the restoration of natural coral reefs (Edwards and Gomez, 2007).

At the coast, one of the most widely applied strategies for enhancement is the addition of features that aim to retain water and mimic natural rock pools to increase biodiversity on artificial structures (Browne and Chapman, 2011; Evans et al., 2015; Firth et al., 2014a). Rock pools at the 10s of mm-m scale have been found to have double the number of species compared to freely draining areas of artificial structures (Moschella et al., 2005). Other methods of ecological enhancement at the coast include the addition of intermediate complexity (mm-scale surface roughness), which promotes early colonising species that are fundamental in community development (Coombes et al., 2015) and retrofitting pits and holes to promote increased biodiversity (Hall et al., 2018).

The potential habitat value of these structures greatly depends on how successful they are at creating a more heterogeneous surface and replicating natural rocky shore features and complexity (Chapman and Bulleri, 2003; Glasby and Connell, 1999). This also includes decisions regarding materials used, such as selecting lighter coloured lithologies for rock armour to reduce the influence of heat retention, as the selection of certain lithologies can influence the development of communities (Green et al., 2012). Darker materials increase substratum temperature, which could exacerbate the effects of a warming climate and increase thermal stress for species compared to those on lighter substratum (Kordas et al., 2014). As such, incorporating microhabitats and associated features that retain water during low tide into the design of seawalls and other coastal infrastructure must be carefully considered, designed and implemented with the continuing conversion of natural to artificial shoreline, as there is no one size fits all solution to enhancing habitat and subsequently biodiversity on artificial structures (Strain et al., 2017b).

1.6 Policy rationale

Ecological enhancement is not yet a mandatory requirement of construction in the UK, but it can mitigate against habitat and biodiversity loss and contribute towards the aspirations of several UK and EU policies. Although very little UK and EU legislation refers directly to ecological enhancement, there are many policies that can be used to implement enhancement strategies at the coast. For example, the Marine Strategy Framework Directive (Directive 2008/56/EC) requires 'Good Environmental Status' of the EUs marine waters to be achieved by 2020, with each member state setting environmental targets. Ecological enhancement could contribute towards this. In addition, both the EC Water Framework Directive (WFD, 2000/60/EC) and the EC Directive on EIA (85/337/EEC) and (97/11/EEC) (e.g. Shaldon scheme (UK), Firth et al., 2014b) are key legal frameworks that can be interpreted to include ecological enhancement. The EC Directive on EIA requires that the environmental consequences of projects are identified and assessed prior to construction or authorisation and so including ecological enhancement at the planning phase could mitigate some of the impacts of construction.

The EC Water Framework Directive (WFD, 2000/60/EC) offers one of the most influential policies as it requires no deterioration of water bodies, including coasts, and to achieve and maintain good ecological and chemical status of all water bodies. This legislation outlines the requirements for heavily modified water bodies (HMWBs), which captures those with coastal defences, to meet 'good ecological potential'. Ecological enhancements of engineered designs could be incorporated to satisfy this requirement (Naylor et al., 2012a).

Comparatively, the Habitats Directive (1992/43/EC) and Birds Directive (1979/409/EC) are less directly leveraged into promoting ecological enhancements since rocky intertidal habitats are not a listed habitat (Naylor et al., 2012a). However, they can be used to ensure no adverse effects on the integrity of sites where construction is occurring and to mitigate some of the long-term negative impacts of construction, such as habitat reductions of prey species of protected bird species (as used in the Hartlepool Headland scheme, Naylor et al., 2017b).

At the UK level, several policies have the potential to support ecological enhancement (listed in Naylor et al., 2012a). The UK Marine Policy Statement (HM Government, 2011) states that development should "aim to avoid harm to marine ecological, biodiversity and geological conservation interests (including geological and morphological features" (2.6.1.3), which lends to both promoting mitigation strategies in the form of enhancement techniques and better understanding geodiversity on intertidal rocky shores in order to replicate these features. Although this statement does not directly advocate for ecological enhancement, its stipulation that coastal developments "may include benefits for marine ecological, biodiversity and geological conservation interests" (2.6.1.4) does favour designs that consider ecological requirements and the replication of microhabitats of importance.

1.7 Thesis structure and aims

At its core, this thesis aims to combine innovative science with real world applications to support policy and industrial applications. The four data chapters in this thesis are developed to achieve this by starting with highlighting the importance of rock material and rock mass properties on natural rocky shores via a coarse scale study of lithology and ecology (Figure 1-2). This work then looks towards developing ecological and biogeomorphological survey methods to increase the understanding of rocky shore biodiversity-geodiversity interactions in order to achieve applied outcomes. This involves drawing from the understanding of lithology and the microhabitat preferences of intertidal species on natural rocky shores, which will contribute to the better understanding of habitat requirements and sampling methodologies on rock coasts. This in turn will further the evidence base for ecological enhancement work, improving the habitat potential of engineered structures through more refined design considerations including material choice, positioning and design, as showcased by the schematic in Figure 1-2. In order to do this, this thesis aims to address several knowledge gaps, identified in Table 1-1.

Table 1-1. Thesis aims and associated chapters.

Aim		**Associated Chapter**
[1]	Examine the importance of including rock material and rock mass properties in ecological studies.	Chapter 2, 3, 5
[2]	Identify the need for more detailed habitat sampling on natural rocky shores.	Chapter 3
[3]	How the spatial distribution of organisms on natural rocky shores relate to rock material and rock mass properties (i.e. microhabitats such as crevices, pools, pits).	Chapter 3
[4]	Provide further insight into biogeomorphic interactions on rock coasts.	Chapter 2, 3
[5]	Extend understanding of the potential for ecological enhancement to enhance biodiversity on structures.	Chapter 4, 5

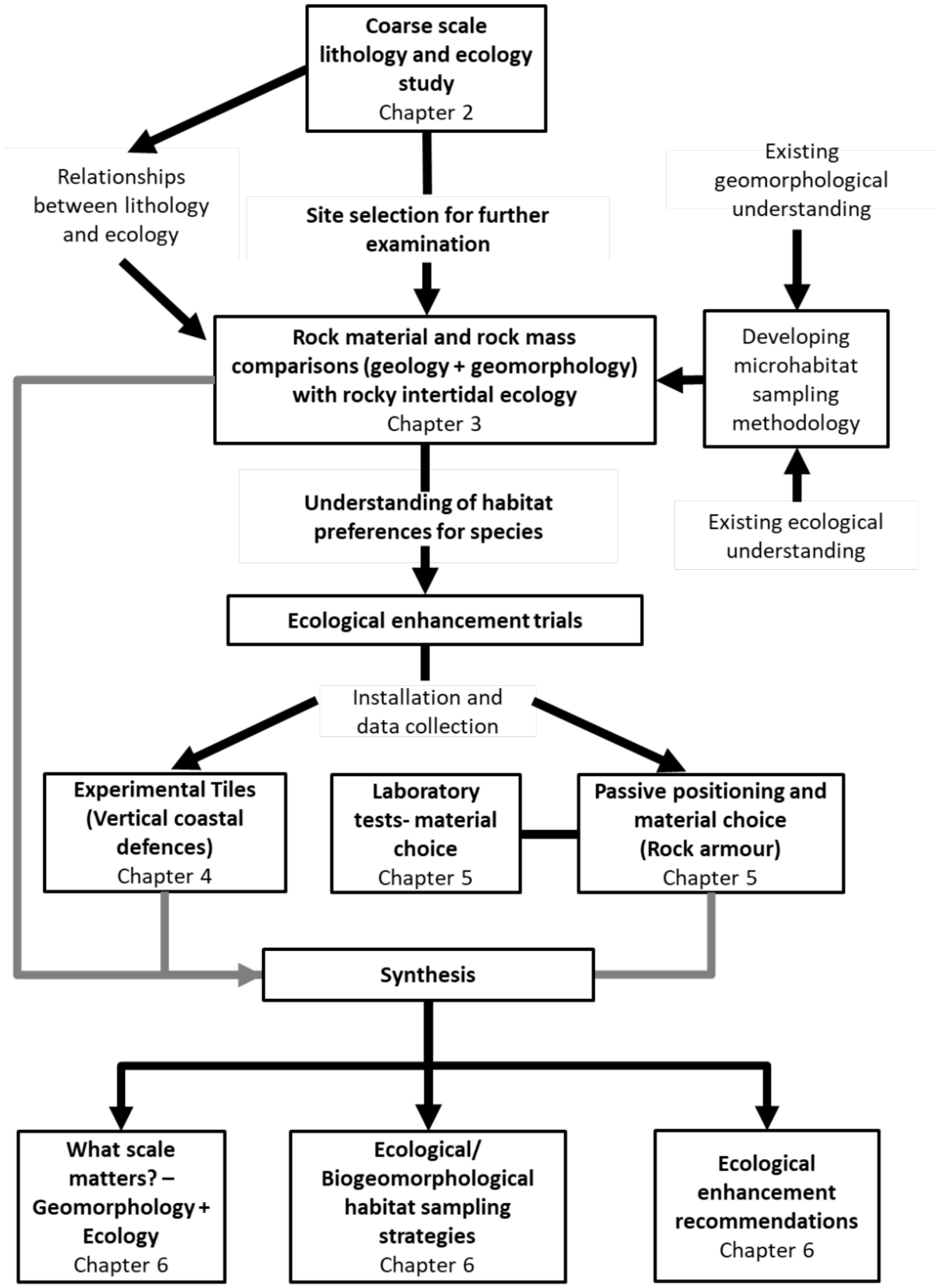

Figure 1-2. Schematic of how chapters feed into each other and overall understanding.

Surveys of natural rocky shores were conducted in the mid-upper intertidal zone in order to directly inform engineering design as often, at lower tidal heights, the influence of microhabitats on species distributions would be lesser as their influence would be blanketed by high cover of algae. Algal cover was found to provide sufficient refuge for intertidal species and has already received more attention in ecological literature than microhabitats (Section 1.2). Conducting surveys in the mid-upper intertidal zone also furthers biogeomorphological understanding. For example, if more species are found to congregate in microhabitats for refuge in the mid-upper intertidal zone, bioerosive species, such as limpets, become points of erosion and drive geomorphic change as their grazing produces fine sediment (Naylor et al., 2012b).

Scale is a running theme throughout this thesis as chapters move from the understanding of biota-substrate interactions on natural shores from the regional (kms-100s of km) to the site scale (10s of m) and on to individual artificial enhancement trials (10s-100s of m) to attract more species to structures. Different spatial scales will have varying influence on ecology. For example, the positive influence of microhabitats on species richness and abundance at the site scale would have a limited influence at the regional scale where other factors such as climate and larval supply may exert more of an influence. In addition to this, the scale of complexity in ecological enhancement work will influence settling species and early colonisers at the mm-scale (Coombes et al., 2015), while cm-dm scale enhancements are more likely to increase species richness and abundance (Firth et al., 2014b; Loke and Todd, 2016; Naylor et al., 2017b). As scale is an important factor to consider, interactions between ecology and lithology and structural features (microhabitats) were measured at the micro-scale of microhabitats, the meso-scale of sites and macro-scale of regions.

This thesis subdivides into three strands that examine interactions across multiple scales (mms-100s of km):

- Strand 1 focuses on the theory behind geodiversity-biodiversity interactions on rocky coasts used to develop Strands 2 and 3. Understanding the interrelationships between biodiversity and geodiversity (Strand 1) is fundamental for the development of ecological and biogeomorphological science and increasing future ecological resilience
- Strand 2 looks at the applications of this understanding in the field, examining rock material and rock mass properties and how they influence species distributions, richness and abundance. This develops a more detailed microhabitat sampling methodology to inform ecological and biogeomorphological understanding and links into Strand 3.

- Strand 3 utilises the understanding gathered from examining natural shore interactions between ecology and geomorphology to improve the multi-functionality of hard coastal defence structures and inform future resilience. This includes making informed decisions regarding surface roughness, microhabitat design (width and depth) and the selection of ecologically informed materials for rock armour. This extends understanding of the potential for ecological enhancement to enhance biodiversity on structures.

Strand 1

Chapter 1. Introduction and Aims

This chapter provides an overview of the research area, namely the understanding between biodiversity and geodiversity on rocky shores and includes a systematic review examining how ecological studies incorporate microhabitats in field surveys. This section provides a brief examination of the influence of lithology and structure on ecology, which is expanded on in depth in Chapters 2 and 3. A policy rationale as to why ecological enhancement is important is also provided, with background information on natural rocky shores and ecological enhancement strategies that set up Chapters 2-6.

Chapter 2. Coarse Scale Regional Analysis of Species Richness and Abundance Data (MARCLIM) and Lithology

This chapter investigates whether a relationship between biology (from broad-scale biodiversity sampling data, i.e. long-term ecological research with broad geographic UK coverage) and lithology (sedimentary, metamorphic and igneous) can be identified using existing datasets. This research aimed to determine whether patterns between species richness and abundance could be estimated based on coarse-scale lithology data (1:10000 or 1:25000 scale on geological maps). If this could be estimated, it would aid in the designation of conservation zones on natural rocky shores based on this understanding. It also allows us to determine which spatial scales geodiversity-biodiversity interactions are most evident and/or important.

Strand 2

Chapter 3. The Importance of Rock Material and Rock Mass Properties and Geomorphic Features-An Examination of Habitat Provision across Rocky Shores

This chapter builds on the understanding that microhabitat features are not as frequently sampled in rocky shore surveys as they should be (Chapter 1) and the findings from Chapter 2 to investigate how rock material and rock mass properties influence biodiversity on several rocky shore sites. The primary focus develops a more detailed sampling method, incorporating width, depth and water holding capacity of small-scale geomorphic features (microhabitats) as measurements in standard ecological quadrat surveys and noting the features that species reside in to make informed conclusions about the importance of these habitats for species. This included a comparison of lithologies (i.e. rock material) between sites on the west coast and east coast of Scotland, with sites selected from the coarse scale sites surveyed in Chapter 2 and a within site comparison of microhabitat utilisation by species. A further site scale comparison was conducted on two adjacent lithologies, their associated geomorphological features and the habitat value of these. Another study was conducted in south Wales, which was selected due to its ease of access while another field campaign was being undertaken and the previous research conducted on the platform (Naylor, 2001). The study in south Wales was conducted on a single shore platform examining the influence of particular microhabitats – crevices, by assessing relationships between jointing density and intertidal species and the associated habitat value of these features. This determined whether species congregated in joints and if so, would further biogeomorphological understanding on the contribution of biology to the weakening of joints via bioerosion. A final aspect to this research created a ‘hotspot’ model of species richness and abundance with regards to the presence of features and water in quadrats on a single shore. This work provides further insight into biogeomorphic interactions on rock coast and the spatial distribution of organisms on natural rocky shores in relation to rock material and rock mass properties (i.e. microhabitats such as crevices, pools, pits). This research develops the understanding of the relationship between geology, geomorphological features and species response on natural rocky shores in order to inform the design of ecological enhancement strategies that better mimic these interactions.

Strand 3

Chapter 4. Ecological Enhancement of Concrete Tiles- Habitat Complexity at the mm-cm Scale

Ecological enhancement trials using concrete tiles are being conducted globally (Strain et al., 2017b). This research was conducted to increase the evidence base for UK enhancement trials on seawalls and is the largest multi-site, multi-design trial of ecologically enhanced concrete tiles in the UK. This chapter evaluates the effectiveness of eight ecologically enhanced designs that varied at the mm-cm scale in their complexity from low (control, clearing) to intermediate (mm-scale surface texture) and high-complexity (microhabitats or relief >10 mm) in their ability to increase species richness and abundance on the surface. Further testing was also conducted to determine whether the presence of microhabitats influenced humidity and temperature on the tiles. This would contribute to the understanding of how the addition of these geomorphological features can help create intertidal refugia on vertical coastal structures.

Chapter 5. Passive Ecological Enhancement- Optimising Ecological Suitability using Rock Material and Rock Mass Properties: the Hartlepool Headland Coastal Defence Scheme

This chapter evaluates the success of passive enhancement strategies, i.e. informed material choice and positioning of rock armour boulders in the intertidal zone, from the mm-dm scale on a live coastal defence scheme at the Hartlepool Headland, north east England and a subsequent comparison with a scheme of similar age at nearby Skinningrove, north east England. The aims were two-fold. One, to determine whether the passive ecological enhancement improved the habitat quality for prey species of internationally important birds being affected by the scheme. Two, to inform future rock armour/revetment design with regards to the most appropriate material choice for ecology and ecosystem engineering using laboratory tests on albedo, porosity and surface roughness to identify the most ecologically suitable lithologies. This includes those with the most ecological engineering potential over the design life of coastal engineering schemes. Field studies were predominately conducted on Shap granite at Hartlepool and compared the ecology on passively positioned (enhanced) boulders with partially enhanced (not positioned but base level of enhancement through ecologically suitable material selection at start of scheme). Subsequently a study was conducted comparing Shap granite with the Norwegian granite at nearby Skinningrove, two of the materials used in laboratory tests, to determine the ecological suitability of each material in the field.

Chapter 6. Synthesis

This chapter synthesises the research and uses the main findings to show how ecological sampling on natural shores and ecological enhancement strategies on artificial shores can be developed. As such it highlights ways in which this research can contribute to geomorphological/biogeomorphological and ecological theory as well as expanding the understanding of the complexity of interactions between biodiversity and geodiversity. The chapter rehearses key findings on the development of ecological sampling methodologies and improvements to the designs of ecological enhancements on artificial shores that draw from an understanding of similar interactions on natural shores. These findings help inform future ecological, biogeomorphological and engineering work on natural and artificial shores.

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Chapter 2. Coarse Scale Regional Analysis of Species Richness and Abundance Data (MARCLIM) and Lithology

2.1 Summary

Coarse scale lithology data were compared with broad scale biological data at the national and regional scale to determine whether a relationship between biology and lithology can be identified from a desk-based study of existing data. Species richness and abundance between sedimentary, metamorphic and igneous lithologies is compared in national and regional scale analyses. Findings indicate that observed differences are more likely the result of differences between locations including exposure and temperature or within site variation in structural components, such as microhabitats, that were not examined in this chapter but are analysed in Chapter 3. These data highlight that coarse scale analyses of lithology coupled with broad-scale biological data is insufficient to determine patterns in species distribution and that lithological patterns may be more prominent at the finer, site scale (10s -100s of m).

2.2 Introduction

The interrelationships between biodiversity and geodiversity are key to improving the management, mitigation and adaptation strategies that are needed for society and ecology to cope with anticipated climate change (Bruneau et al., 2011). Patterns in community richness and abundance can vary across multiple scales of resolution from national scale (100s of kms) where both differences in climate and changes in wave exposure may exert an influence and at the scale of individual shores (Thompson et al., 1996). The geomorphological processes operating at the coast range from the micro-scale (μm-mm) to the national scale (100s of km). These processes are fundamental in supporting ecosystems, habitats and the subsequent ability of species to adapt to environmental and anthropogenic pressures (Bruneau et al., 2011). In the uncertainty of a changing climate, it is fundamental that these underpinning relationships are understood across a range of spatial scales and that any proposed management approaches are spatially integrated at the landscape and ecosystem scale.

Rock material properties including surface roughness, hardness, chemical composition, porosity and colour are known factors that influence the settlement of macro- and microscopic species on rocky shores (Holmes et al., 1997) and can influence the geographical range of species. It is expected, based on previous geological research, that sedimentary lithologies would be more porous, softer and have greater calcium content than metamorphic and igneous lithologies (Coombes et al., 2011; Coombes and Naylor, 2012; Sempere-Valverde et al., 2018), which makes these lithologies more ecologically suitable.

Surface roughness can influence small-scale (mm-scale) variations in biofilm and cyanobacterial cover (Hutchinson et al., 2006), with biofilm shown to be an important ecosystem engineer for some lithologies (Coombes et al., 2011) and in providing biochemical cues for species colonisation and recruitment (Coombes et al., 2015). Both primary productivity and diatom abundance have been shown to increase with increased mm-scale surface roughness (Sempere-Valverde et al., 2018). Higher substratum roughness can also aid in reducing desiccation stress. Work on sandstone with micro- surface features (<2 mm) found lower thermal and hydric stress which may have contributed to algal attachment and growth (Sempere-Valverde et al., 2018).

Calcareous lithologies, such as limestone with a high calcium carbonate content, can influence the overall community composition of flora species, at least in the terrestrial environment (English Nature, 2004). Softer lithologies can be more friable and at risk from erosion on exposed shorelines, which may increase the mortality rates of settling species such as barnacles (Herbert and Hawkins, 2006). However, softer, calcium-rich lithologies can also offer greater habitat potential as bioerosive species are able to act as ecosystem engineers increasing the surface roughness of the rock and increasing porosity and potential water pooling at the micro-scale (µm–mm) (Coombes et al., 2011; Coombes and Naylor, 2012; Naylor et al., 2012b). This increased surface roughness provides a more suitable surface for the recruitment of other intertidal organisms (Coombes et al., 2011) and the associated higher porosity reduces potential desiccation stress, increasing ecological suitability for intertidal species as the rock remains wetter for longer across the low tide period (Coombes et al., 2011; Green et al., 2012). Calcareous substrates were also found to have higher diversity than siliceous substrates in the lower intertidal (Ramos et al., 2016). However, this may not be a definitive relationship as the effects of lithology will vary with specific species and Ramos et al.

(2016) sampled only two siliceous shores. In addition, many species in the intertidal zone have calcium carbonate shells, with the availability of carbonate influencing biomineralisation by species (Fitzer et al., 2015). However, it is noted that both climate and relief may be more influential in varying communities than the calcium content of lithologies (English Nature, 2004).

The colour of rocks can also influence community composition as lighter coloured artificial substratum was found to reduce thermal stress for colonising species due to having lower rock surface temperatures (Kordas et al., 2014), with darker colours additionally having been shown to slow the growth of Ulva sporelings and delay germination (Finlay et al., 2008). For some species, such as spirorbid tubeworms, colour was a determinant of recruitment to boulders with darker boulders preferred over lighter coloured boulders, regardless of lithology (James and Underwood, 1994) but several other factors including temperature of the underside of boulders, surface texture and differences in chemical composition were noted to potentially influence these differences (James and Underwood, 1994).

Topographic complexity exerts an influence on ecological communities and is known to influence species richness and abundance. Johnson et al. (2003) found that habitat heterogeneity can exert an influence on species richness at the regional scale (between regions) but did not vary between shores within a region. Most studies examine topographic complexity on less-than-whole-shore spatial scales, but the underlying effects of complexity may bleed through to differences amongst sites.

With these factors in mind, it is expected that rock material properties, driven by lithology, should influence intertidal biota as a result of differences in its aforementioned characteristics and composition. This chapter uses EDINA Digimap (© Geological Map Data BGS © NERC 2018) geological data. However, with the coarse scale of the geological dataset and the variation in lithological properties such as roughness, porosity, colour, chemistry (even within the same lithology), it is difficult to examine several of these properties at the scale of this study due to the lack of information available. As such, only general lithological patterns were examined (sedimentary, metamorphic and igneous) and calcium content was estimated into three distinct groupings (low, intermediate and high) based on existing composition information for several lithologies found in Scotland and south Wales (BGS, 2018a, BGS, 2018b).

In the UK, a lack of spatially extensive datasets for rocky intertidal organisms led to the MarClim project, set up in 2001, to improve the understanding of the impacts of climate change over the long term and how this might affect coastal species diversity and distribution (Laffoley et al., 2005). The MarClim data (Burrows et al., 2017) consists of annual ecological surveys carried out at over 100 sites around the UK coastline. These surveys record species abundance measured on the 'SACFOR' scale (Super-abundant, Abundant, Common, Frequent, Occasional and Rare). Two series of surveys were conducted, with the UK-wide rocky shore site series from 2002-2014 (Scottish sites 2002-2010) and the second series of surveys conducted all along the Scottish rocky coastline from 2014/2015. As noted by Burrows et al. (2017), 111 of the 156 sites surveyed in 2014/2015 in Scotland were matched or within 2 km of sites visited in the 2002-2010 surveys. The 2014/2015 data extended the spatial and temporal coverage of the initial surveys. Whilst these data have been used to examine the changes in abundance and distribution of species surveyed in the context of recent temperature-based climate change (Burrows et al., 2017), no consideration was given to how the underlying lithology of each site may influence species occurrence or abundance.

As the nature of the underlying lithology of an area influences the distribution of both habitats and species (English Nature, 2004), this thesis aimed to assess whether lithological control was evident within the existing MarClim dataset. The existing MarClim dataset provides extensive spatial and temporal data with the ability to objectively analyse species patterns in relation to the underlying lithology. This aids in the development of management or mitigation strategies and allows an insight into whether certain lithologies have a greater ecosystem service value in terms of biodiversity and community functioning, and therefore need greater protection. This study focused on examining the influence of lithology on ecology, with structural components, such as microhabitats, analysed in Chapter 3 as they could not be identified from a desk-based study using existing data.

For this chapter, EDINA Digimap geological data was combined with the MarClim datasets to answer the following research question: Can we predict the ecological suitability of lithologies based on existing broad scale ecology and coarse scale lithological data?

To answer this, the following three hypotheses were examined:

1) Sedimentary lithologies will have greater species richness and abundance than metamorphic and igneous lithologies.
2) Community composition will differ between lithologies in the same region.
3) Higher calcium lithologies (limestone) will have greater species richness and abundance than lower calcium lithologies.

2.3 Methods

2.3.1 MarClim sampling strategy

The MarClim data provides an extensive broad-scale, skeleton analysis of ecological data on rocky shores with areas identified for sampling being up to or exceeding 100 m in extent. These areas were walked, and a checklist of species used for timed surveys, typically up to 30 minutes, with species then allocated a SACFOR category. Few quadrat counts were used to aid in categorising species (SACFOR). The apparent absence of species was also recorded (MarClim, 2008).

2.3.2 Data analysis methods

The 2014 and 2015 dataset is cited as Burrows et al. (2017). The 2002-2014 UK data is not publicly available but can be request from the Marine Biological Association (Marine Biological Association, 2019). For species abundance data, the SACFOR method was converted to numbers so that integer numbers represented each category, 0 for absent, 1 for rare, 2 for occasional up to 6 for super abundant. Some species were not recorded but could be present and were classed as NA as their absence is the result of a negative result in the search.

The MarClim SACFOR abundance data was then converted to species richness based on the presence of species. This meant that mobile, algae/lichen, barnacle, sessile fauna (e.g. anemones, mussels, oysters, specified for each analysis), lower shore and total species richness could be extracted from the abundance data. Barnacles are a key species throughout this thesis due to their global abundance, notable role as ecosystem engineers and bioprotective capacity (Coombes et al., 2015; 2017), and so were analysed individually. Other sessile fauna (anemones, mussels and oysters) were also analysed individually as their habitat requirements help in developing more ecologically suitable habitats on coastal defences. Ascidians, hydroids, sponges, *Sabellaria* sp., *Polychaeta* spp. and bryozoans were grouped together as these species are not typically associated with the mid-upper intertidal zone, the main area of interest of this thesis. These species were analysed for the national comparison and classified as 'other' for analyses. For the regional analysis, these species were only included for total richness and abundance as their limited occurrence prevented individual analysis.

The research conducted in this chapter is divided into a national study using the 2014/2015 Scotland dataset (Figure 2-1) and a series of regional analyses using the 2002-2014 dataset. Regions for analysis were extracted from the MarClim dataset to establish a broad spatial spread to support subsequent field surveys, with sites in the east and west coast of the central belt in Scotland, alongside sites in the south of Wales (Figure 2-1). These regions were selected for their ease of access for subsequent field studies (Chapter 3) to examine site scale differences in lithology and structure. Individual site geologies were identified using the DiGMapGB-50 Rock Unit on EDINA Digimap (© Geological Map Data BGS © NERC 2018). GPS locations of the MarClim sites were co-located with local site lithology at the 1:10000 or 1:25000 scale. A total of 46 lithologies were identified within the three areas selected from the 2002-2014 MarClim data, and subsequently grouped into sedimentary, metamorphic and igneous lithologies for analysis with species richness data.

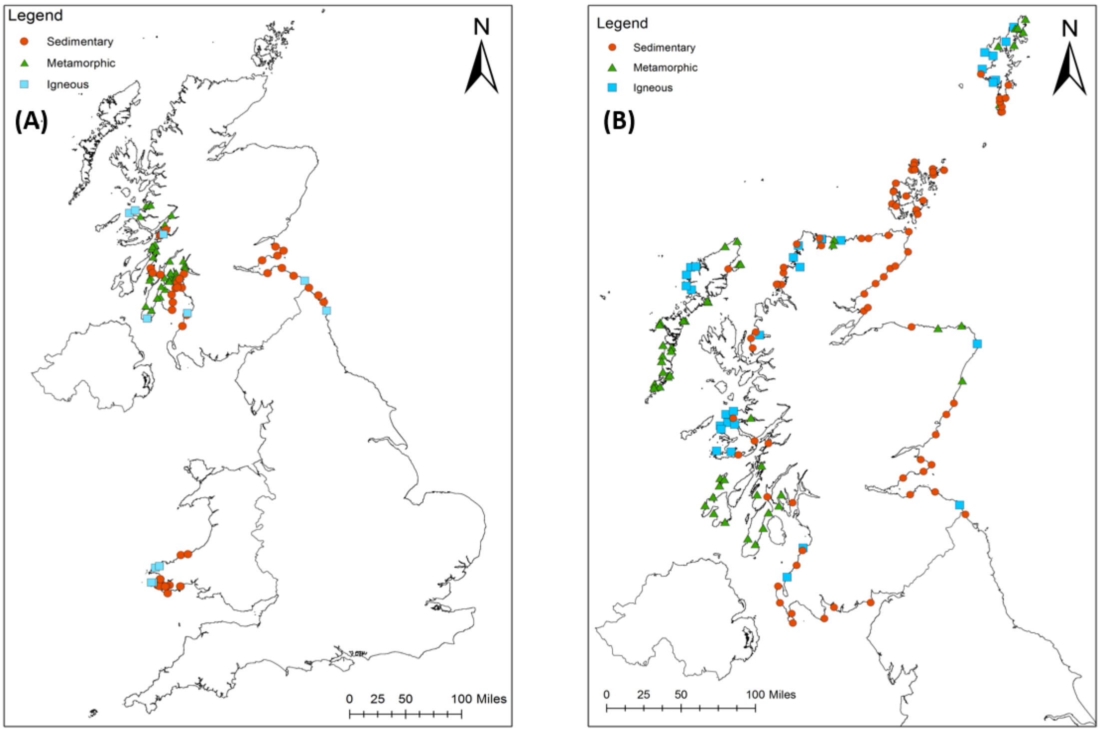

Figure 2-1. Map of UK MarClim survey sites used in the (A) regional biodiversity-lithology analyses and (B) national analyses with sedimentary, metamorphic and igneous lithologies at sites marked.

Further to this, the average calcium content of lithologies was compared with ecological data by grouping their calcium content into percentage range bands (BGS, 2018a), as outlined in Table 2-1. For lithologies where mineralogy was not specified at the level of lithology or rock unit (BGS, 2018a), lithologies were grouped higher if they contain calcium-based compounds such as calcium silicate or calcite minerals (BGS, 2018b). Where carbonate rocks, categorised as having >50% calcium carbonate (Ford and Williams, 2007) were found mixed with sandstone or other lithologies, they were categorised as medium calcium content. Due to the lack of exact values for each lithology, this represents a coarse, but acceptable, level of analysis.

Table 2-1. Calcium content groupings and associated lithologies.

Group	1 (low)	2 (intermediate)	3 (high)
Calcium based (C %)	0≤ C%<25	25≤ C%<50	50≤ C%< 100
Associated lithologies	All other lithologies (combination of sedimentary, metamorphic and igneous)	[Mudstone, Sandstone, Limestone] [Semipelite, Calcareous] [Sandstone + Nodular Limestone] [Pelite + Subordinate Metalimestone] [Limestone, Sandstone, Siltstone and Mudstone] [Sandstone, Siltstone, Dolomitic Limestone]	[Limestone] [Metalimestone] [Limestone + (Subequal/Subordinate) Argillaceous Rocks], Interbedded]

2.3.3 Statistical Methods

For both datasets (2002-2014 regional and 2014-2015 Scotland), Quasi-Poisson and Negative Binomial Generalised Llinear Models (GLMs) were carried out to detect significant relationships between species richness and abundance variables and lithology. GLMs were selected due to their flexibility in analysing non-normal distributions and counteracting the influence of the high volume of zero observations typical of count data, even using the SACFOR scale. Generalised linear hypothesis testing (GLHT) was then used with Tukey's for multiple pairwise comparisons.

For the 2002-2014 regional data, initial comparisons using GLMs condensed lithology into igneous and non-igneous for cross-comparison between regions, as the greatest differences were expected between igneous and non-igneous lithologies, with non-igneous thought to be more ecologically suitable. This initial analysis had to be grouped as metamorphic lithologies were not sampled in ecological surveys in the east of Scotland or in south Wales. Differences were detected between the east of Scotland and south of Wales and further GLMs highlighted that there were differences between regions for several variables and so regions were subsequently analysed individually. Quasi-Poisson and Negative Binomial GLMs were then conducted to compare between sedimentary, metamorphic and igneous lithologies for ecological variables in each region to determine if specific lithologies affected species richness and abundance.

To assess the similarity of ecological communities between lithologies, ANOSIM (analysis of similarities) and SIMPER, which calculates dissimilarity between groups using Bray-Curtis dissimilarity, were used to examine variation in species abundance within each region. For community analysis, where NAs (species not recorded but could be present at site) were present in 90% or more of samples, species were excluded from the dataset. If species were present in more than 90% of samples then NAs in the data were recoded back as zeros, although it should be noted that this may influence the results of the analysis. Following this, species were grouped by trophic level and linear models were conducted with trophic level as a predictor.

Following the initial analysis for each dataset, richness and abundance was compared between calcium groupings (low, intermediate, high) using further GLMs with the methods outlined above. For all GLMs, both the residual deviance and Akaike Information Criterion (AIC) (where available) was examined to determine the most appropriate model to use (Zuur et al., 2009). All analyses were conducted using R version 3.5.1 (R Development Core Team 2018).

2.4 Results

2.4.1 National study – Scottish lithology (MarClim 2014/2015)

The national study included 223 surveys conducted across 156 sites in Scotland, subdivided into sites with sedimentary (n=94), metamorphic (n=85) and igneous (n=44) lithologies, with a total of 93 species recorded across the surveys. Although there was no significant difference in the total species richness between lithologies, examination of species groupings identified patterns. Sedimentary lithologies were significantly higher than metamorphic lithologies in attracting greater mobile species richness and abundance ($p<0.001$, Table 2-2, Figure 2-2). This was then compared with an interaction between lithology and coasts (west, north and east) to determine whether this effect was influenced by region. Differences in mobile species richness and abundance between lithologies was not influenced by coast but in excluding the effect of lithology, the west coast had lower mobile abundance than the north and east coast ($p<0.05$, Table 2-2). In general, metamorphic lithologies had lower richness and abundance, even where results were not statistically significant (Figure 2-2). A full table of statistical results is available in Table A 2-1.

Table 2-2. Post-hoc comparisons (with GLM specified) results for significant species richness and abundance metrics between a/b) lithological type and c) coast (***= p<0.001, **=p<0.01, *=p<0.05, NS= Not significant).

a) Mobile species richness- Quasi-Poisson				
Comparison	**Estimate**	**Std. Error**	**Z value**	**P-value**
Metamorphic- Igneous	-0.119	0.053	-2.266	NS
Sedimentary- Igneous	0.041	0.050	0.823	NS
Sedimentary-Metamorphic	0.161	0.043	3.781	***
b) Mobile abundance - Negative Binomial				
Comparison	**Estimate**	**Std. Error**	**Z value**	**P-value**
Metamorphic- Igneous	-0.186	0.057	-3.271	**
Sedimentary- Igneous	0.103	0.054	1.901	NS
Sedimentary-Metamorphic	0.290	0.046	6.331	***
c) Mobile abundance ~ Coast -Negative Binomial (excluding lithology)				
Comparison	**Estimate**	**Std. Error**	**Z value**	**P-value**
North-East	0.037	0.072	0.508	NS
West-East	-0.181	0.069	-2.634	*
West-North	-0.217	0.046	-4.713	***

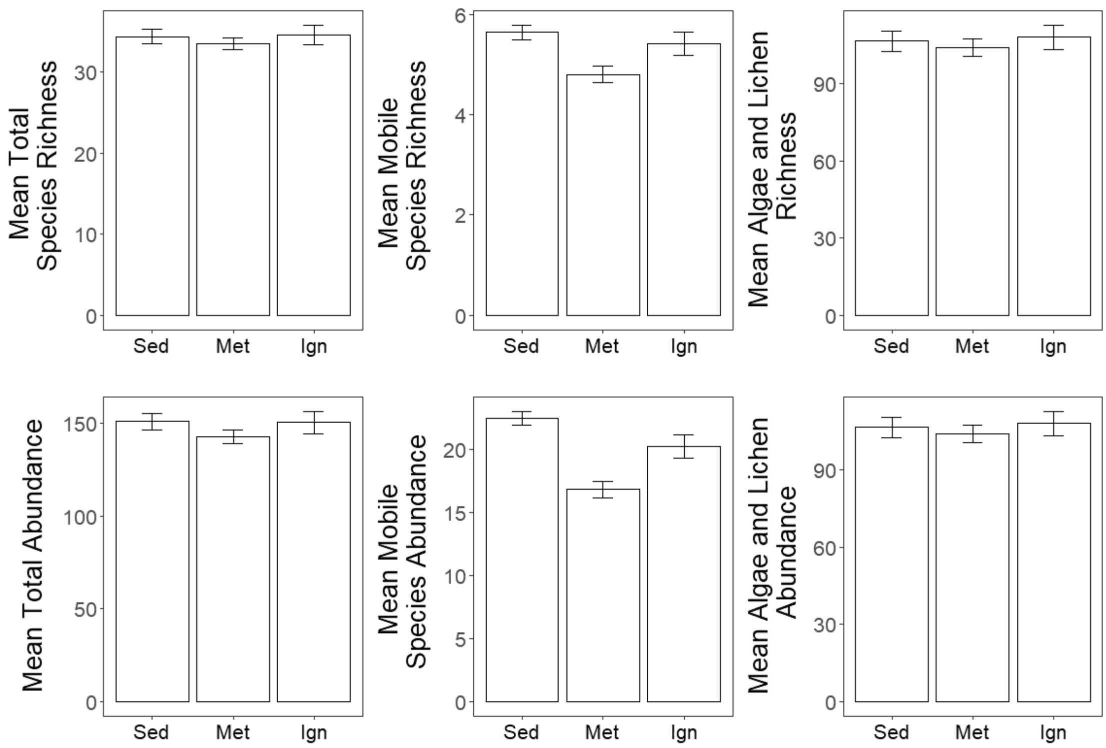

Figure 2-2. Mean total, mobile and algae and lichen richness and abundance comparisons between lithologies in Scotland (n=94 sedimentary, n= 85 metamorphic and n=44 igneous) ($\overline{x}$ ± standard error).

2.4.2 National study- calcium content

Lithologies were reclassified according to their calcium content which resulted in most lithologies falling into the low calcium (0≤ C%<25) category (n=216). Only 7 surveys were conducted on intermediate and high calcium lithologies (n=4 and n=3 respectively). Algae and lichen richness and abundance, total abundance and sessile fauna abundance was found to be higher on low and intermediate compared to high calcium lithologies ($p<0.05$,Table 2-3,Figure 2-3) and total species richness was also greater on intermediate than high calcium lithologies ($p<0.05$,Table 2-3). Barnacle abundance and sessile fauna richness did not statistically differ between calcium groupings but were higher on average on low and intermediate lithologies than those with high calcium (Figure 2-3). This indicates that there is another factor that may override the effects of lithology, such as surface process dynamics or structural controls (examined in Chapter 3) that would influence the suitability of surfaces and habitats.

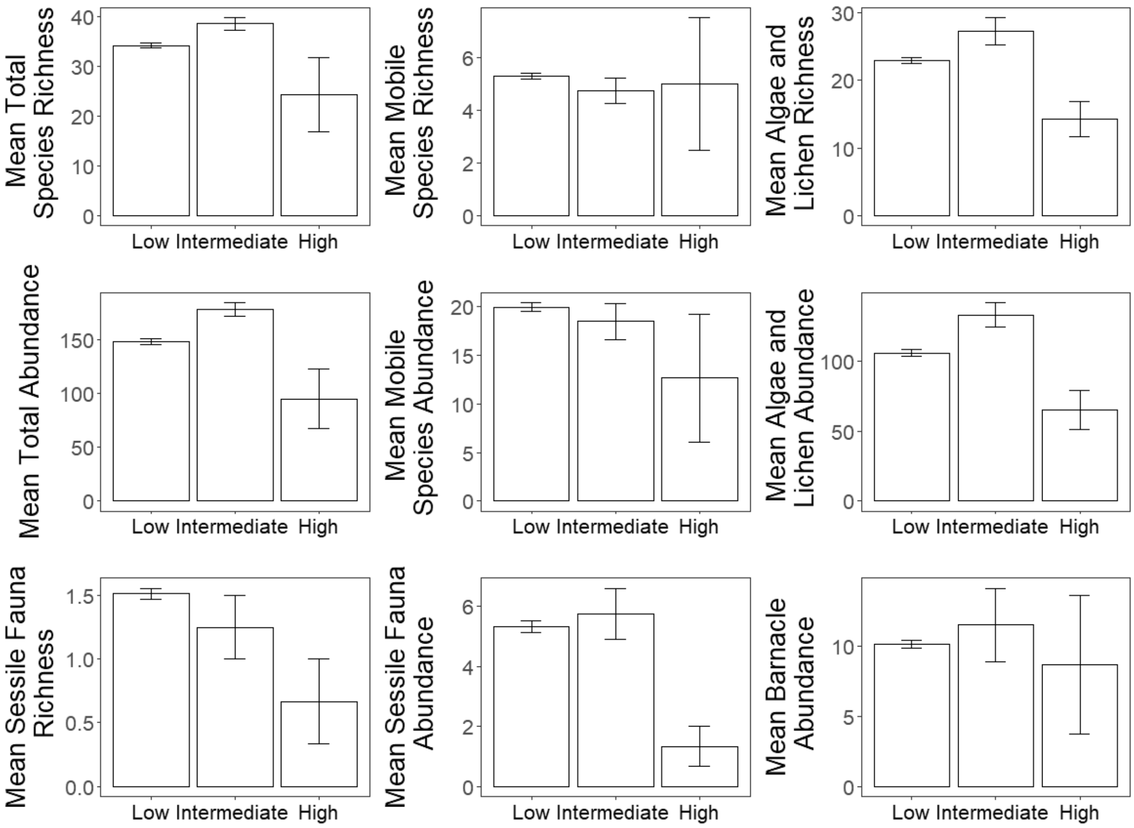

Figure 2-3. Mean species richness and abundance for ecological metrics by calcium content (low, intermediate and high) ($\bar{x}$ ± standard error).

Table 2-3. Significant results from post-hoc comparisons of calcium content with species richness and abundance (***= p<0.001, **=p<0.01, *=p<0.05, NS= Not significant).

a) Total species richness- Negative Binomial				
Comparison	**Estimate**	**Std. Error**	**Z-value**	**P- value**
Intermediate-High	0.459	0.188	2.437	*
Low-High	0.339	0.151	2.251	NS
Low-Intermediate	-0.120	0.115	-1.039	NS
b) Algae and lichen richness- Negative Binomial				
Comparison	**Estimate**	**Std. Error**	**Z-value**	**P- value**
Intermediate-High	0.643	0.238	2.699	*
Low-High	0.471	0.194	2.430	*
Low-Intermediate	-0.172	0.141	-1.218	NS
c) Total abundance - Negative Binomial				
Comparison	**Estimate**	**Std. Error**	**Z-value**	**P- value**
Intermediate-High	0.628	0.210	2.991	**
Low-High	0.443	0.162	2.734	*
Low-Intermediate	-0.185	0.136	-1.360	NS
d) Algae and lichen abundance - Negative Binomial				
Comparison	**Estimate**	**Std. Error**	**Z-value**	**P- value**
Intermediate-High	0.716	0.262	2.737	*
Low-High	0.487	0.202	2.411	*
Low-Intermediate	-0.229	0.170	-1.353	NS
e) Sessile fauna abundance- Negative Binomial				
Comparison	**Estimate**	**Std. Error**	**Z-value**	**P- value**
Intermediate-High	1.462	0.588	2.487	*
Low-High	1.385	0.530	2.614	*
Low-Intermediate	-0.076	0.259	-0.294	NS

2.4.3 Summary- National scale

At a national scale of analysis, metamorphic lithologies appear to be the least suitable for intertidal ecology, with lower mobile species richness and abundance than sedimentary and igneous lithologies. The general pattern across most ecological metrics of species richness and abundance highlighted that high calcium lithologies may not necessarily be the most ecologically suitable.

2.5 Regional study

206 surveys were analysed (n=101 sedimentary, n=72 metamorphic and n=33 igneous) across the east and west of Scotland and the south of Wales. Metamorphic lithologies had lower species richness on average than sedimentary and igneous lithologies (Table 2-4), with sedimentary lithologies recording the greatest species richness (43 species). No metamorphic sites were surveyed in the east of Scotland and south of Wales (Table 2-4).

Table 2-4. Summary statistics of total species richness by lithology for combined regional data and the number (n) of surveys/sites of each lithology by location.

	Mean total species richness	Std. Error	r	Min	Max	East Scotland (number of sites)	West Scotland (number of sites)	South Wales (number of sites)
Sedimentary	22.33	0.78	101	8	43	10	27	64
Metamorphic	14.14	0.38	72	9	23	0	72	0
Igneous	23.67	1.15	33	8	32	2	5	26

For total and mobile species richness, the south of Wales was significantly greater than the east of Scotland ($p<0.001$,Table A 2-2), with this pattern also being observed with sessile fauna richness ($p<0.05$) and barnacle species richness (maximum of 6 species recorded), which also differed with the west of Scotland ($p<0.01$). No differences were observed for species richness when grouping sedimentary and metamorphic (non-igneous) and comparing them with igneous lithologies.

There were also regional differences in total and mobile species abundance ($p<0.001$), although grouping lithology exerted no influence over species abundance at this scale (Table A 2-3). Barnacle abundance, which was analysed separately due to the preference of barnacles for intermediate surface roughness (mm-scale), did not differ between grouped lithologies but was significantly different between regions ($p<0.05$,Table A 2-3).

In comparing between regions, the largest differences were observed between the south of Wales and the two Scottish sites, with total, mobile and sessile fauna species richness significantly higher on the Welsh coast than on both Scottish coasts ($p<0.001$, Table 2-5, Figure 2-4). This was also true for total, mobile and sessile fauna abundance (p<0.001, Figure 2-4). Differences were additionally observed between the west and east of Scotland for mobile species richness and abundance, sessile fauna abundance and barnacle abundance (p<0.05, Table 2-5). Algae and lichen species richness and abundance was greater in the south of Wales than the west coast of Scotland (p<0.01) and barnacle abundance differed between all regions (p<0.001).

Table 2-5. GLM (Quasi-Poisson and Negative Binomial) for species richness and abundance between the three examined regions (***= p<0.001, **=p<0.01, *=p<0.05, NS= Not significant).

a) Total species richness- Quasi-Poisson				
Comparison	**Estimate**	**Std. Error**	**Z-value**	**P-value**
South Wales- East Scotland	0.661	0.074	8.909	***
West Scotland- East Scotland	0.026	0.076	0.338	NS
West Scotland- South Wales	-0.636	0.031	-20.795	***
b) Mobile species richness- Quasi-Poisson				
Comparison	**Estimate**	**Std. Error**	**Z-value**	**P-value**
South Wales- East Scotland	1.130	0.084	13.465	***
West Scotland- East Scotland	0.280	0.086	3.273	**
West Scotland- South Wales	-0.850	0.030	-28.652	***
c) Algae and lichen species richness- Quasi-Poisson				
Comparison	**Estimate**	**Std. Error**	**Z-value**	**P-value**
South Wales- East Scotland	0.152	0.103	1.476	NS
West Scotland- East Scotland	-0.214	0.105	-2.043	NS
West Scotland- South Wales	-0.366	0.050	-7.377	***
d) Sessile fauna richness (Mussels, Anemones, Oysters) - Quasi-Poisson				
Comparison	**Estimate**	**Std. Error**	**Z-value**	**P-value**
South Wales- East Scotland	0.946	0.178	5.317	***
West Scotland- East Scotland	0.313	0.181	1.730	NS
West Scotland- South Wales	-0.633	0.064	-9.891	***
e) Total abundance- Negative Binomial				
Comparison	**Estimate**	**Std. Error**	**Z-value**	**P-value**
South Wales- East Scotland	0.539	0.075	7.214	***
West Scotland- East Scotland	0.014	0.075	0.193	NS
West Scotland- South Wales	-0.52	0.034	-15.420	***
f) Mobile abundance- Negative Binomial				
Comparison	**Estimate**	**Std. Error**	**Z-value**	**P-value**
South Wales- East Scotland	1.052	0.092	11.450	***
West Scotland- East Scotland	0.229	0.093	2.452	*
West Scotland- South Wales	-0.823	0.035	-23.454	***
g) Algae and lichen abundance- Negative Binomial				
Comparison	**Estimate**	**Std. Error**	**Z-value**	**P-value**
South Wales- East Scotland	-0.026	0.130	-0.198	NS
West Scotland- East Scotland	-0.222	0.129	-1.728	NS
West Scotland- South Wales	-0.197	0.061	-3.209	**

h) Sessile fauna abundance- (Mussels, Anemones, Oysters)- Negative Binomial				
Comparison	**Estimate**	**Std. Error**	**Z-value**	**P-value**
South Wales- East Scotland	1.075	0.202	5.327	***
West Scotland- East Scotland	0.598	0.203	2.949	**
West Scotland- South Wales	-0.477	0.073	-6.563	***
i) Barnacle abundance - Quasi-Poisson				
Comparison	**Estimate**	**Std. Error**	**Z-value**	**P-value**
South Wales- East Scotland	1.108	0.109	10.210	***
West Scotland- East Scotland	0.468	0.110	4.258	***
West Scotland- South Wales	-0.640	0.036	-17.665	***

In individually examining the three regions, lithology appears not to exert a significant influence on any metric of species richness or abundance on the east coast of Scotland or the west coast of Scotland. However, despite the lack of significance, sedimentary lithologies attracted greater richness and abundance of species than igneous lithologies on the east coast (Figure 2-4, Figure 2-5), with an average of 14.3 (±1.04 SE) total species richness compared to 11.5 (± 2.5 SE) on igneous lithologies and a total abundance of 57.5 (±3.68 SE) compared to 44 (± 6 SE). A similar pattern was observed on the west coast with sedimentary lithologies the most ecologically suitable of the three lithologies, even if results were not significant (Figure 2-4, Figure 2-5). In south Wales, only sessile fauna abundance (anemones, mussels, oyster) was significantly different, with sedimentary lithologies having a greater abundance than igneous ($z=4.56$, $p<0.001$).

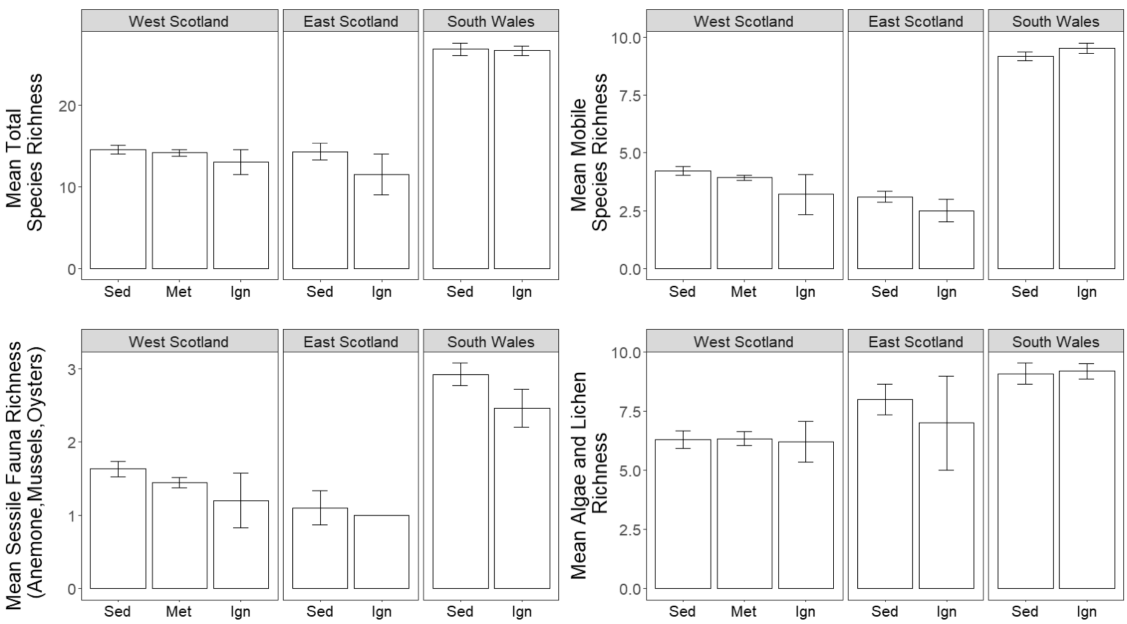

Figure 2-4. Mean total, mobile, sessile fauna and algae and lichen richness between lithologies (sedimentary, metamorphic and igneous) in each region ($\bar{x}$ ± standard error).

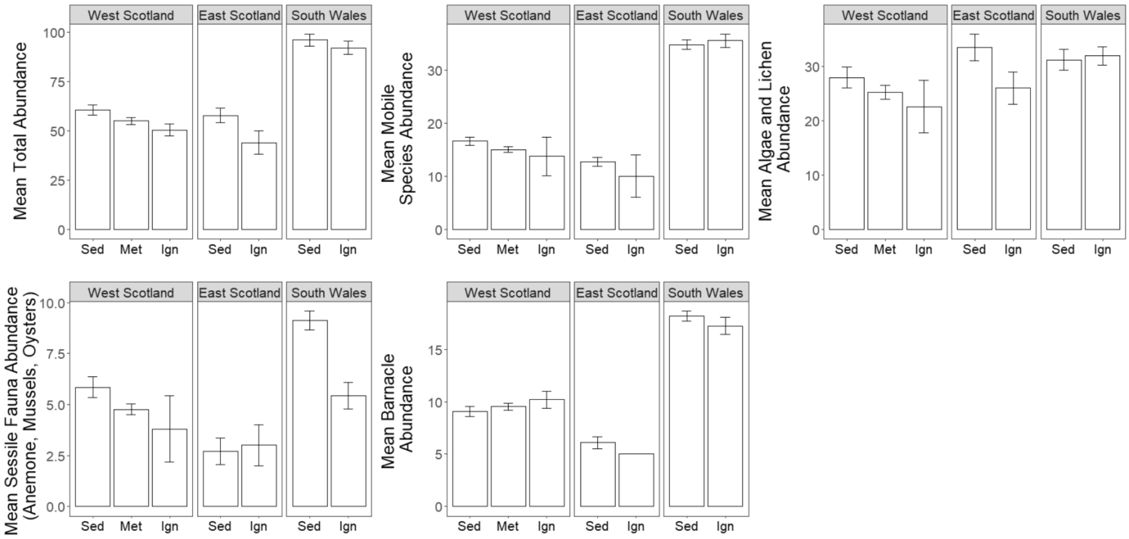

Figure 2-5. Mean total, mobile, algae and lichen, sessile fauna and barnacle abundance between lithologies (sedimentary, metamorphic and igneous) in each region ($\bar{x}$ ± standard error).

The results of ANOSIM also showed that ecological communities on contrasting lithologies did not significantly differ (east coast Global R=0.300, p=0.151; west coast Global R=0.03, p=0.262; south Wales Global R=-0.006, p=0.517).

On the east coast of Scotland 29 species were recorded on sedimentary shores (n=10) against 16 on igneous shores (n=2), with the 12 most influential species accounting for 74.6% of the dissimilarity between samples (Table A 2-4), with sedimentary and igneous lithologies 40.5% different from each other. *Semibalanus balanaoides* was equally the abundant sessile species on both lithologies. After this, the most abundant sessile species (fauna and flora) on sedimentary shores were *Fucus vesiculosus, Fucus spiralis, Fucus serratus* and *Mastocarpus stellatus* compared to *Fucus spiralis, Laminaria digitata, Pelvetia canaliculata* and *Mastocarpus stellatus* on igneous shores (Table A 2-4). *Patella vulgata* and *Littorina littorea* were the most abundant mobile grazers on both shores. In grouping species by trophic level, producer (algal) species were no more abundant on sedimentary shores than on igneous shores (Table 2-6).

On the west coast of Scotland, there were limited differences in communities between lithologies when comparing metamorphic and sedimentary (36.62%), metamorphic and igneous (40.53%) and sedimentary and igneous (41.08%) lithologies. Between metamorphic and sedimentary lithologies, the 14 most influential species accounted for 71.36% of the differences (Table A 2-5), this was similar to metamorphic and igneous comparisons (n=13 species for 71.7% of differences, Table A 2-6) and sedimentary and igneous comparisons (14 species for 73.46% of differences, Table A 2-7). *Semibalanus balanoides* was the most abundant species on all three lithologies. The most abundant sessile species were *Pelvetia canaliculata, Fucus serratus* and *Fucus spiralis* in varying order on all three lithologies, with *Laminaria digitata* also in high abundance on igneous shores. *Patella vulgata, Littorina littorea* and *Nucella lapillus* were the mobile grazers found in highest abundances on each lithology (Table A 2-5, Table A 2-6, Table A 2-7). When species were grouped by trophic level, there were no differences between trophic levels with sedimentary and igneous comparisons and metamorphic and sedimentary comparisons. However, there appears to be a greater abundance of large brown algae on sedimentary than igneous shores, as exemplified by ratios for large brown algae >1 (Table A 2-7). Mobile grazers seem generally less abundant on igneous shores. Splitting species into trophic level, linear models show that there is some support for plants being more abundant on sedimentary shores ($p<0.05$) (Table 2-6). Further analysis was undertaken combining grazers and predators to compare with plants, but this was not significant (Table 2-6).

In south Wales, the 20 most influential species contributed to 70.1% of the differences between sedimentary and igneous lithologies that were 31.71% different. Species of barnacle were amongst the most abundant species on both sedimentary and igneous shores (*Semibalanus balanoides* and *Chthamalus montagui*). Mobile grazers were in high abundance on both lithologies, particularly *Patella vulgata,* and *Fucus serratus* was the most abundant algae (Table A 2-8). There was no difference between trophic levels when comparing between lithologies (Table 2-6).

Table 2-6. Linear models of ratios of species abundance between lithologies with trophic level as a factor (***= p<0.001, **=p<0.01, *=p<0.05, NS= Not significant).

East coast- sedimentary vs igneous				
	Estimate	**Std. Error**	**T value**	**P value**
(Intercept)	1.170	0.217	5.407	***
Trophic level 2	-0.541	0.319	-1.698	NS
Trophic level 3	0.275	0.612	0.449	NS
West coast- metamorphic vs sedimentary				
	Estimate	**Std. Error**	**T value**	**P value**
(Intercept)	0.858	0.084	10.167	***
Trophic level 2	-0.137	0.119	-1.150	NS
Trophic level 3	0.360	0.239	1.507	NS
West coast- metamorphic vs igneous				
	Estimate	**Std. Error**	**T value**	**P value**
(Intercept)	0.928	0.098	9.417	***
Trophic level 2	-0.271	0.139	-1.949	NS
Trophic level 3	0.316	0.289	1.134	NS
West coast- sedimentary vs igneous				
	Estimate	**Std. Error**	**T value**	**P value**
(Intercept)	0.986	0.095	10.332	***
Trophic level 2	-0.273	0.133	-2.063	*
Trophic level 3	0.195	0.261	0.747	NS
	Estimate	**Std. Error**	**T value**	**P value**
(Intercept)	0.771	0.090	8.614	***
Plant	0.215	0.134	1.606	NS
Wales- sedimentary vs igneous				
	Estimate	**Std. Error**	**T value**	**P value**
(Intercept)	0.981	0.079	12.368	***
Trophic level 2	0.098	0.106	0.924	NS
Trophic level 3	0.142	0.194	0.733	NS

2.5.1 Species richness- calcium content

Dividing lithology by calcium content showed only a few sites in each region with intermediate-high calcium content (Table 2-7). This reduces the ability to draw definitive conclusions on patterns between species richness and calcium content in south Wales and the west of Scotland as most sites here were categorised as of low calcium content (Table 2-7). Species richness and abundance did not significantly differ with calcium content across most comparisons. Sessile fauna richness and abundance on the east coast of Scotland (predominantly *Actinia equina* and *Mytilus edulis*) was greater on lithologies with lower rather than intermediate calcium content (p<0.05, Table 2-8) and barnacle species richness on the west coast was greater on lithologies with high calcium content (limestones) than with low calcium content <25% (p<0.05, Table 2-8).

Table 2-7. N numbers for sites within each region according to grouping by calcium content.

Calcium group		East Scotland	West Scotland	South Wales
1 (0≤ C%<25)	Low	8	90	89
2 (25≤ C%<50)	Intermediate	4	9	0
3 (50≤ C%< 100)	High	0	5	1

Table 2-8. Quasi-Poisson GLM results for significant comparisons of calcium and a) sessile fauna richness and b) sessile fauna abundance on the east coast and post-hoc comparisons for b) barnacle abundance on the west coast (***= p<0.001, **=p<0.01, *=p<0.05, NS= Not significant).

	East coast			
a) Sessile fauna richness - Quasi-Poisson				
	Estimate	**Std. Error**	**T value**	**P-value**
(Intercept)	0.319	0.175	1.821	NS
Intermediate calcium	-1.012	0.446	-2.269	*
b) Sessile fauna abundance- Quasi-Poisson				
	Estimate	**Std. Error**	**T value**	**P-value**
(Intercept)	1.288	0.176	7.339	***
Intermediate calcium	-1.288	0.504	-2.555	*
	West coast			
a) Barnacle abundance - Quasi-Poisson				
	Estimate	**Std. Error**	**Z value**	**P-value**
High- Low	0.267	0.114	2.335	*
Intermediate- Low	0.137	0.092	1.491	NS
Intermediate- High	-0.128	0.140	-0.914	NS

2.5.2 Regional Summary

There is clear variation between regions, but the effects of lithology are difficult to extract at this scale. Although not significant, the general pattern highlights that sedimentary lithologies potentially attract higher richness and abundance of species at both the east and west coast of Scotland, with more variable results on the welsh coast. Community composition was similar between lithologies with no dominant community of plants or animal species at any of the examined sites. There is little affiliation between higher richness and abundance of species and calcium content of lithologies at this spatial scale.

2.6 Discussion

Any variation observed between lithologies is hard to pinpoint as specifically being a result of lithology because of the variety of characteristics that could not be examined at this spatial scale due to a lack of sufficient data. This includes further rock material characteristics including surface roughness, chemistry, albedo and porosity for specific lithologies (e.g. limestone, sandstone, semipelite) as well as rock structural/mass properties that may influence the geomorphological processes, features and microhabitats of rocky shores. Although mobile species richness and abundance was greater on sedimentary over metamorphic lithologies in the national scale study, mobile abundance was found to be influenced by the coast, with the west coast found to have lower abundance than the north and east coasts when excluding the effect of lithology.

The east and west coast of Scotland had lower species richness and abundance compared to south Wales for all species groupings in the regional analysis, but lithology exerted no significant influence within the east or west coast, highlighting that region had more of a significant influence on richness and abundance. This leads to the rejection of hypothesis 1, that sedimentary lithologies will have greater species richness than metamorphic and igneous lithologies as this was not statistically true for either study. Hypothesis 2, that community composition would differ between lithologies in the same region, can be rejected as for all comparisons of lithology there was no significant variation in community. It appears at this coarse level of analysis that lithology does not influence community composition as dominant sessile and mobile assemblages were relatively consistent between lithologies in each region.

Observed variation between lithologies, such as sessile fauna abundance being greater on sedimentary than igneous lithologies in south Wales and mobile species richness being greater on sedimentary than metamorphic lithologies in the national scale study may be the result of other factors including rock material properties, such as porosity or surface roughness, or rock mass properties, such as variability in jointing, exerting an influence on community composition (Raimondi, 1988; Schoch and Dethier, 1996). Differences between locations, including wave exposure, water temperature (Schoch and Dethier, 1996), planktonic food supply and suspended sediment loads would exert more of an influence on community composition than lithology at the observed spatial scale (100s of kms). The influence of lithology may be more relevant at smaller spatial scales, but this broad scale analysis does not reveal this effect. This is in agreement with previous findings that highlight lithology cannot be considered a major driver of regional scale rocky shore species richness and diversity (Burrows et al., 2014; Ramos et al., 2016).

Calcium content was additionally found not to be a primary driver in influencing species richness or abundance at this scale. Low and intermediate calcium contents performed better in the examined ecological variables than high calcium content lithologies, even where results were not significant. The exception to this was on the west coast in the regional analysis, barnacle abundance was greater on high calcium over low calcium lithologies but this may be due to the higher surface roughness of limestones resulting from susceptibility to weathering and erosion, including ecosystem engineering, (Coombes et al., 2011) and barnacle preference for intermediate (mm-scale) surface roughness (Coombes et al., 2015; Herbert and Hawkins, 2006). These findings result in the rejection of hypothesis 3, that higher calcium lithologies (limestone) will have greater species richness and abundance than lithologies with moderate and low calcium. However, the lack of intermediate-high calcium sites in both studies reduces the ability to draw conclusions at this scale as to whether higher calcium content favours greater species richness and abundance.

Previous reports have noted that local scale variation in rock mass properties, the geomorphological features created and the microhabitats these make (e.g. cracks, crevices) may be more important in influencing species than any variability in rock chemistry (English Nature, 2004). It is likely that the few differences observed between lithologies at this scale of study are more likely a reflection of structural effects, through variation in habitat complexity and diversity (Frost et al., 2005; Burrows et al., 2014) as the intertidal zone is composed of a multitude of habitat types that are not considered by this scale of study. In addition, the scale and speed of the MarClim data means that boring species are overlooked as they are difficult to find in coarse scale surveys and these species are commonly found on softer, calcium-rich lithologies (Naylor et al., 2012b; Pinn et al., 2008).

2.7 Conclusion

Using lithology as a predictor of species distribution and associated richness and abundance at the regional scale is problematic, as the settlement and recruitment potential of different lithologies is highly variable. Lithology, and associated rock mass and material properties related to it, are likely an important determinant of species richness and abundance but not at a regional scale. The MarClim data provides an extensive broad-scale analysis of ecological data from rocky shores across the UK but due to the scope and timed nature of the original MarClim surveys, there was limited ability to examine species richness and abundance within habitat features, which would tease apart the relative effects of structure (rock mass properties) and lithology (rock material properties). More detailed ecological data would also have been more likely to record key rock boring species. The results of this regional scale study highlight that lithology may be more a modifier of patterns than a driver. Lithology appears to be less important than rock mass structure and it is now crucial to examine how structure modifies patterns in species richness to better understand the interrelationships between geodiversity and biodiversity. As a result of these findings, the next chapter examines field studies undertaken to examine the influence of lithology, rock mass structure, the geomorphic features that result (e.g. rock pools) and the effects of these on ecological response at multiple field sites. Several field sites used were repeat surveys of MarClim sites to better account for microhabitats such as cracks, crevices and pools that can influence species distribution by providing refuge during the stressful low tide period.

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Chapter 3. The Importance of Rock Material and Rock Mass Properties and Geomorphic Features-An Examination of Habitat Provision across Rocky Shores

3.1 Summary

This work examines the influence of lithology and different types of small-scale geomorphic features (microhabitats) found on rocky coasts such as pools, pits, cracks, crevices and ledges on rocky shore biodiversity (species richness and abundance). These interactions were examined through four studies: (1) a comparative study of rock material and rock mass properties at four sites in Scotland to establish the role of lithology and microhabitat type on ecology, (2) a comparison of two lithologies (limestone and sandstone) at the same site to identify the importance of lithology and microhabitat type on ecology (3) the creation of a model of rocky shore species density in relation to microhabitat density and water retention in microhabitats by integrating Terrestrial Laser Scans of geomorphic complexity overlain with ecological variables (species richness and mobile species abundance) (4) a study of jointing density at the Glamorgan coast, comparing the effects of low and high density jointing (crevices and cracks) on species richness and abundance within the same bed layer. Species richness and abundance was found to significantly differ with lithology and the presence of microhabitats on each shore, with deep pools being the optimal habitat type with regards to these ecological variables. The higher abundance of species in pools and crevices (depending on site) highlights that the presence of larger-scale features such as pools, crevices and ledges are important for the survival of intertidal species, as crevices can function as refuge from predators and wave stress and pools can reduce desiccation stress at low tide. This greater provision of habitat through increased surface area, increased protection and greater variety of surface topography suggests that complex lithologies and associated microhabitats are more likely to have higher species richness and abundance, particularly when compared to adjacent areas of shore platforms that lack this geomorphic complexity.

3.2 Introduction

Examining the influence of lithology using coarse-scale data and high-level shore-scale summary biological data in Chapter 2 indicated that there was limited effect of rock material properties at this scale. The MarClim surveys examined in Chapter 2 were not structured to look at individual small-scale geomorphic features, i.e. the physical space in which an organism resides (synonymous with microhabitats), as they integrate across all microhabitats, presenting an overview of what was present at each site. This targeted species search approach may reduce the impact of individual lithologies by reporting abundance from sections of rock coasts containing the appropriate habitats for particular species, irrespective of the frequency of the feature.

It is known that lithology and texture (surface roughness) influences water holding capacity (Coombes et al., 2011), large-scale (landform) geomorphic features (Coombes, 2014) and the types of microhabitats that form (Jackson, 2015). The rock material properties influence how the rock will weather and erode, with lithology providing the resisting force as some lithologies are more physically resistant to these processes than others (Naylor, 2005). Structural features on shore platforms influence the types of processes occurring (Trenhaile, 1987). These rock mass properties function as planes of weakness where the action of physical, chemical and biological processes actively makes geomorphic features, such as crevices and pools over time (Chapter 1.1). Where lithologies have similar structures but one is softer and more chemically suitable, such as limestone, it would erode faster and processes like bioerosion (e.g. pitting by limpets) would be more prevalent, which in turn can produce microhabitats such as pits and enhance pool morphology over time (Scheffers et al., 2012). With a combination of these processes, the ecological suitability of the lithology would increase over time as topographic complexity would increase. This interplay between rock material and rock mass properties and environmental processes drives the formation of geomorphic features (microhabitats).

This research addresses the interactions between geomorphic feature types (microhabitats) and ecology within individual shores. Both lithology (rock material properties) and its associated structural features (rock mass properties- microhabitats) influence the profile of shore platforms (Bird, 2011). These were examined to identify the ecological responses to these geodiversity parameters within sites and between sites, to assess consistency of ecological responses to these geodiversity parameters between shores. Hereafter, we refer to this as geomorphic (or topographic) complexity throughout this thesis.

Topographic complexity at a given location is highly dependent on geodiversity, and in turn, this geodiversity acts as an important mechanism in maintaining biodiversity and delivering key ecosystem services (Gordon et al., 2012; Hjort et al., 2015; Londoño-Cruz et al., 2014). Habitat complexity, the number and diversity of habitat types, which is analogous to beta diversity, can result from geodiversity, biogeomorphic interactions (e.g. where organisms actively create topographic complexity in rocks, Naylor et al., 2012b) and by biology itself creating habitat. For example, habitats created by living organisms (termed bioconstruction, Naylor et al., 2002), or in the space between live organisms or inside dead organisms such as barnacles and mussels (e.g. Coombes et al., 2015). Geomorphic and habitat complexity are highly important on rocky shores as environmental variables including temperature, humidity and hydrodynamic exposure have been found to vary in response to the presence of these geomorphic features (Helmuth and Hofmann, 2001; Jackson, 2010).

Physical factors such as substrate type, orientation, thermal stress, wave exposure and elevation directly affect community composition, interactions and distributions, contributing to known patterns of intertidal zonation (Gonor and Kemp, 1978; Harley and Helmuth, 2003; Konar et al., 2016; Schoch and Dethier, 1996). Topographic complexity and resulting microhabitats can create pronounced gradients of environmental conditions from the outside edge towards the inside of habitat features, resulting in unique microclimates (Harley and Helmuth, 2003; Healy, 1996; Jackson, 2015; Meager et al., 2011). Species have been found to settle in pits, pools and crevices (Davidson and Grupe, 2015; Hall et al., 2018; Raffaelli and Hughes, 1978) as these features have been found to provide more humid and less wave exposed habitats than platform surfaces, consequently buffering environmental stress during periods of low tide (Cartwright and Williams, 2012; Jackson, 2010; Lee and Li, 2013; Rickards and Boulding, 2015).

High levels of topographic complexity exert a strong influence over species distribution, richness and abundance as different species have different habitat requirements (Coombes et al., 2011; Helmuth and Hofmann, 2001; Jackson, 2015). Small-scale heterogeneity (<20 cm), which is typically the scale of individual microhabitats, has been found to influence species abundance whereas heterogeneity at the km-scale influences diversity, with richness tending to increase with increased substrate heterogeneity

(Archambault and Bourget, 1996), which is analogous to the effect of beta diversity. Both mobile and sessile intertidal organisms have often demonstrated higher abundances and associations with topographic features in their habitat (Jackson, 2010; Raffaelli and Hughes, 1978), with crevice size and availability having been shown to influence population structure, particularly in increasing the density of Littorinid species (Raffaelli and Hughes, 1978).

Regarding the understanding of marine biodiversity and geodiversity interactions, there is a lack of a standardised methodology for the quantification and measurement of microhabitats and the species within them. The heterogeneous nature of rock coasts and the range of features available, including cracks, crevices, overhangs, ledges, pits and rockpools means that sampling within topographic features is technically difficult and sampling units, such as quadrats, are often positioned away from these features (Bulleri et al., 2005; Menge et al., 2010; Murray et al., 2001; Pister, 2009).

Defining geomorphic features is complex as different disciplines and different authors have varying definitions when it comes to terminology and the width and depth requirements for categorisation. The term fracture in geological literature applies to a range of discontinuities (Palmström, 2001) and is used by rock coast geomorphologists (e.g. Naylor and Stephenson, 2010); this includes features defined as crevices in ecological literature. In addition, few sources define the dimensions required for categorisation. For example, Palmström (2001) in characterising rock mass jointing only noted the length of features rather than width and depth measurements, with micro-cracks stated as less than 0.01 m and cracks ranging from 0.01-1 m in length.

In considering the terminology used throughout this thesis, ecological, geomorphological and geological/rock mechanics literature was considered. For the smallest discontinuities, the term hairline crack was used, typically being mms in length (Goudie, 2004), with these micro-scale features typically very flat with little width and depth compared to their length (Simmons and Richter, 1976) and are often the first appearance of discontinuities within the rock (Higgins and Coates, 1990). These features were noted where they were present as this fine texture may still provide some habitat benefit to species that are known to prefer mm-scale texture, such as barnacles (Coombes et al., 2015).

Increasing the spatial scale, cracks and crevices had the most variation in their definitions and size classifications (Bergeron and Bourget, 1986; Healy, 1996; Raffaelli and Hughes, 1978) (Figure 3-1). Only cracks and crevices are included in Figure 3-1 as these microhabitats had the most disputed definitions of their dimensions when examining ecological literature, whereas other features, such as pools and pits, have a more consistent understanding of what these features are. Few studies offer a quantitative classification when describing or defining topographic features such as cracks and crevices, such as Cartwright and Williams (2012), which defined cracks and crevices by their being too small or large enough for littorinids to fit into. Several sources used ratios to define crevices, with Jackson (2010) stating that crevices were defined by a length: width ratio greater than 3:1, width: depth less than 3:1 and width and depth greater than 1 cm along the entire length and no standing water. Strain et al. (2017b) defined a crevice as an “intertidal or subtidal depression with a length to width ratio > 3:1, and a depth of >1 mm” (2018, p.429). With the extensive nature of the sampling in this research and the aim to determine optimal habitat types for species, it was decided that ratios were not a suitable strategy for definition and that the definition provided by Strain et al. (2017b) could be further divided as this encompassed cracks and crevices. In analysing a range of definitions (Figure 3-1), it was determined that crevices would be defined by their widths exceeding or equal to 2.5 cm and a minimum depth of 2 cm but generally equalling or exceeding the width of the crevice opening. A width of between 1 and 2.5 cm would be defined as a crack and depths had to exceed crack width at opening (Figure 3-1). Any smaller than this and the feature would be defined as a hairline crack, which fell between 1 and 9 mm wide and 1 and 20 mm deep for this research (n=24 hairline cracks total). From personal observations, surveys of several rocky shores and analysis of existing literature, a range of crack and crevice size definitions (width and depth in cm) is included in Figure 3-1.

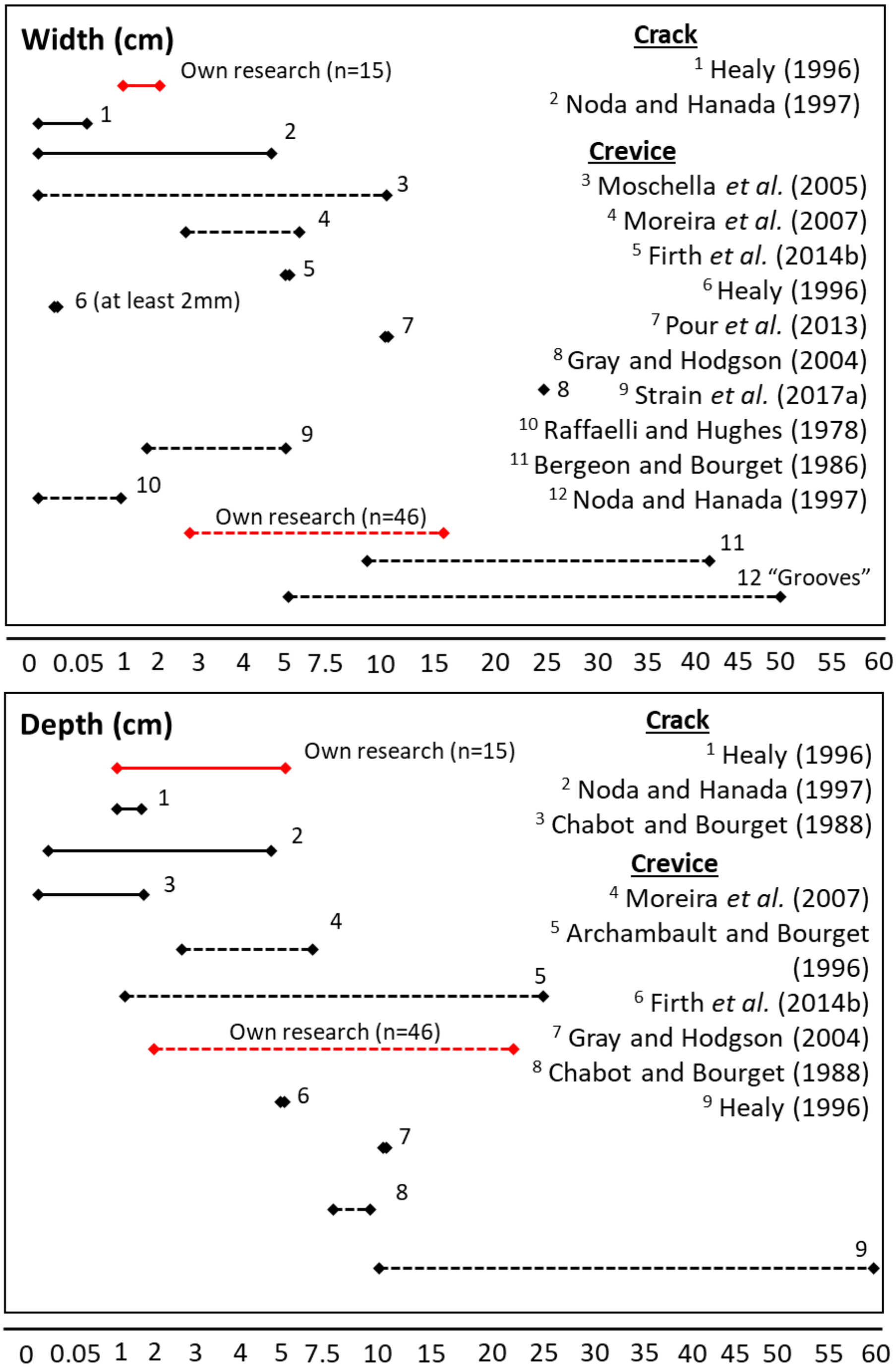

Figure 3-1. Width and depth of cracks and crevices (cm) from ecological literature where feature measurements were quantitatively stated. Solid lines indicate crack measurements and dashed lines are crevices. Red lines are measurements from quadrat surveys conducted in this chapter from n=15 cracks and n=46 crevices.

The presence of standing water in crevices did not qualify the features to be classified as rock pools (Jackson, 2010), instead the water holding capacity (% cover within feature) was noted for every feature. Where standing water was the predominant feature within a depression in the rock, the feature could be defined as rock pool. Rock pools held variable amounts of water at low tide and so were subdivided into deep and shallow pools, with shallow pools having a maximum depth of 2.5 cm. Width varied with these features, shallow pools were frequently shallow depressions that held water on the rock that did not have side height high enough to define them as pits, which were defined as small circular or near-circular depressions on the surface (Trudgill, 1988a). Overhangs were defined as the inwards extension of a hollow 'roof' from the undercutting of a section of the rock, typically under ledges, which are raised or protruding from the shore platform. Table 3-1 shows the full range of widths/depths for each feature recorded in the following studies to showcase the definitions used in this research and associated imagery for features can be seen in Figure 3-2.

Table 3-1. Dimensions of features (width and depth) and number of each feature sampled. Depth is synonymous with height for ledges.

Feature	Width range (cm)	Depth range (cm)	N
Hairline crack	0.1-0.9	0.1-2	24
Crack	1-2	1-5.2	15
Crevice	2.75-15	2-22	46
Pit	2.5-5	1.2-5	14
Shallow pool	2.8-48	0.5-2.5	42
Deep pool	7-100	2.8-24	43
Ledge	9-150	3.1-70	74
Overhang	17-80	3.5-35	5

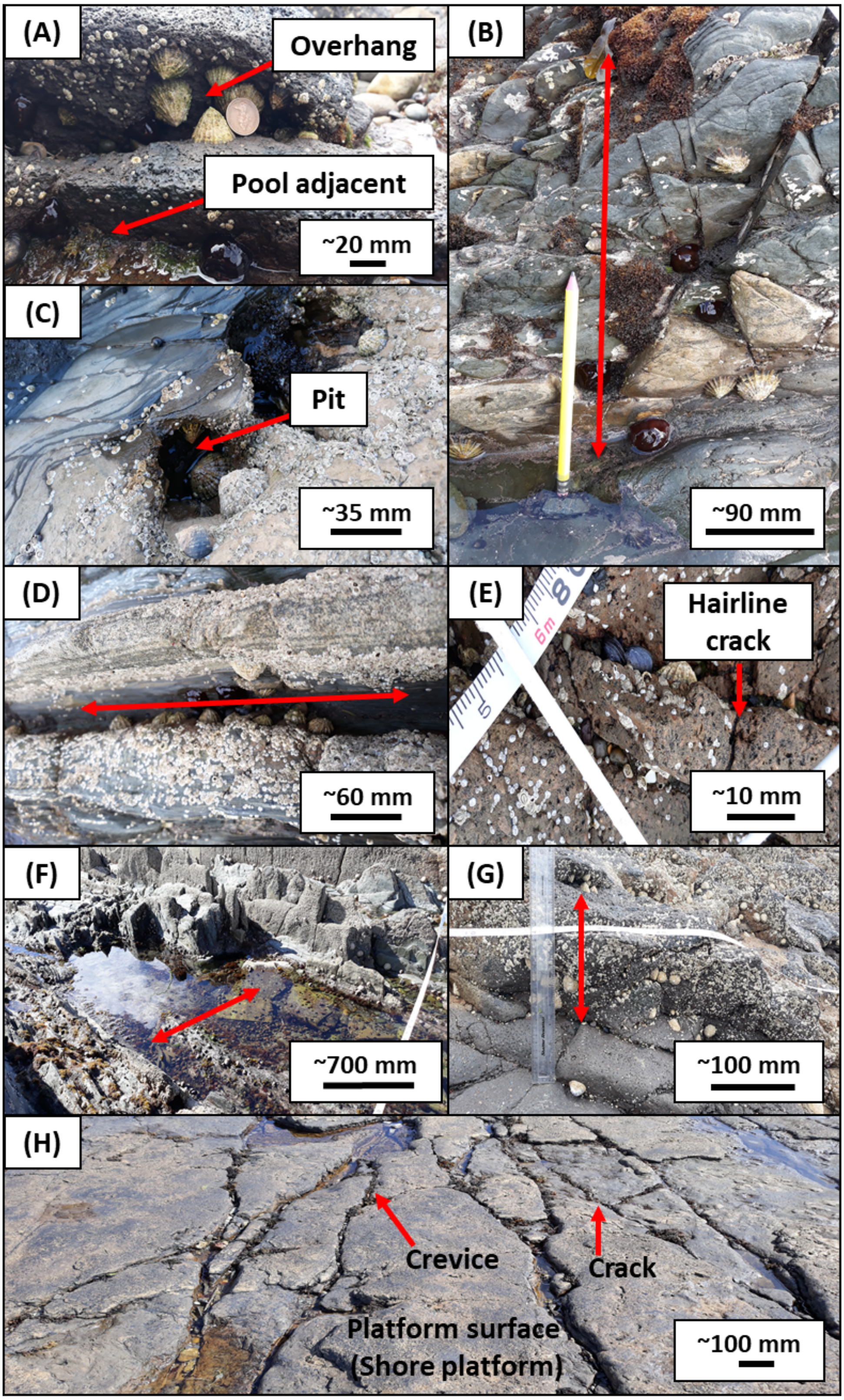

Figure 3-2. Field photographs of (A) an overhang and pool adjacent habitat, (B) ledge habitat with pencil for scale, (C) pits, (D) crevice (E) hairline crack (F) deep pool (G) another ledge and (H) crack and crevice habitat on the platform surface. Red arrows highlight the specific feature. Approximate scale bars (mm) are provided.

Several key geological and geomorphological factors that could modify the distribution patterns of species within and between shores include the nature of the substratum, water retention and presence of microhabitats (Lewis, 1964; Hill et al., 1998). Tidal height and vertical zonation were kept constant by only conducting surveys within the mid-upper intertidal zone in order to focus on the effects of complexity (Chapter 1.3). This zone is where the effects of sea level rise induced coastal squeeze would have the greatest impact where rocky shores are backed by steeper inclines or hard coastal structures, the intertidal zone will decrease (Jackson and McIlvenny, 2011). This research aims to relate back to informing engineering design of coastal defences as structures in the mid-upper intertidal zone typically have lower abundances and diversity of species than lower heights (Firth et al., 2012), with habitat disproportionately lost in this zone (Dugan et al., 2008) and so these structures would benefit from more ecologically informed designs.

This research was developed with the core aim of developing the understanding of biological and physical interactions on rocky shores through the development of a detailed ecological survey methodology to better account for geomorphological features and related rock mass properties. This allowed for comparisons with the adjacent shore platform surface to quantitatively evaluate the importance of habitat complexity for rocky shore species.

As a result, several studies were conducted on multiple shores across Scotland and at one site on the Glamorgan coast, in south Wales, UK (Figure 3-3).

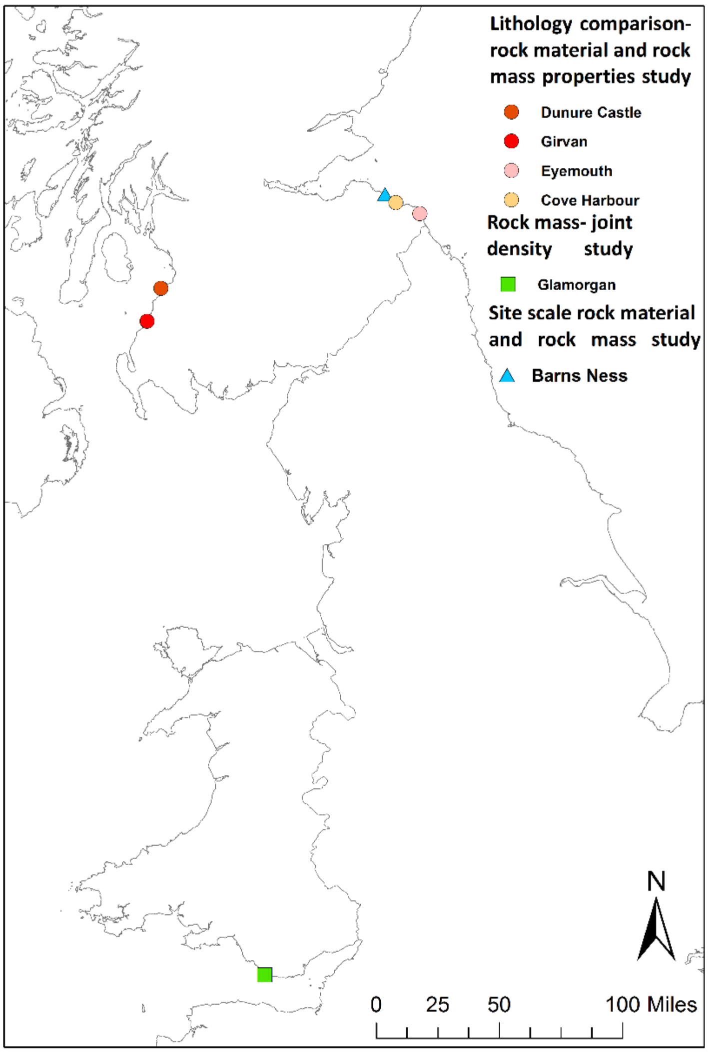

Figure 3-3. Location of field sites for geomorphic rock mass and rock material studies.

Each study (n=4) presents a detailed site-based investigation into the importance of rock mass and rock material properties, identifying which microhabitats support the greatest species richness and abundance. Several research questions were investigated in each study to better understand this.

(1) Do rock material properties influence the presence and abundance of rocky intertidal species?
(2) Do geomorphic and rock mass related features (i.e. microhabitats) increase species richness and abundance?
(3) Which type(s) of microhabitats provide the most important for each common rocky intertidal species?

To answer each question, multiple hypotheses were developed for each individual study building on the findings from Chapter 2 (Table 3-2). These questions thus sought to evaluate if site scale geological and geomorphological factors play a role in the spatial distribution of species and their richness and abundance. The coarse scale of the MarClim data (Chapter 2) resulted in lithology having little effect on species richness and abundance at this level. Adding in the features provided a more detailed survey whilst still comparing between lithologies at the site scale (Study 1).

For Study 2, a site-scale rock mass and rock material study was conducted at Barns Ness (Table 3-2). Two platforms of different lithologies were situated within 50 metres of each other, offering a comparable field site under similar environmental conditions to examine how the rock material properties exert an influence on the biology in addition to examining the influence of microhabitats on the two contrasting lithologies. As noted in Chapter 1 (Section 1.2 and Section 1.5), species on darker lithologies more susceptible to the effects of warming. Barnacles and herbivores have been found to be less abundant and fare poorly on black plates compared to white plates in material colour studies, with black plates found to increase in situ substratum temperature (Kordas et al., 2014). Lighter lithologies with higher surface roughness can reduce thermal stress and encourage early colonising species, influencing the survival and distribution of species on the shore (Coombes et al., 2015; McGreevy, 1985). Higher porosity and water absorption capacity keeps rocks wetter for longer, which is important in reducing desiccation stress during low tide (Coombes and Naylor, 2012). A series of laboratory tests (albedo, porosity, water absorption capacity and surface roughness) were conducted on samples of the sandstone and limestone to better understand the baseline characteristics of the rock material properties of the shore platform. Each platform was then compared for differences between lithologies and within platform differences between microhabitat types.

Study 3 was conducted on the sandstone at Barns Ness that integrated differential GPS (DGPS), terrestrial laser scanning (TLS) and ecological data as this allowed spatially explicit small-scale sampling relative to geomorphic features and created a visualisation of species distribution patters in relation to microhabitat distribution. The resulting model showed the interactions between geomorphic complexity and biotic response.

Finally, to examine the influence of joint density, as crevices are important intertidal habitats (Chapter 1.3), Study 4 was conducted within the same bed layer at Glamorgan in south Wales. This examined whether a more densely jointed survey area (estimated visually and confirmed after fieldwork) had greater species richness and abundance than a less densely jointed area within the same bed layer. This research aimed to highlight the significance of discontinuities play in increasing species abundance (Table 3-2).

Table 3-2. Study overview, research questions (RQ), associated hypotheses and the sources that contributed to their development.

Study	RQ	Hypotheses	Key sources that helped to inform hypotheses
(1) Regional scale (between and within regions) and site scale lithology	**1**	**[H1]** Softer lithologies (limestone and sandstone) will have greater species richness, abundance (mobile and sessile species) and diversity than harder lithologies (basalt and andesite)	(Coombes et al., 2011; Coombes and Naylor, 2012; Firth et al., 2012; McGreevy, 1985; Naylor et al., 2012b; Sempere-Valverde et al., 2018)
comparison- rock material and rock mass properties. Four	**1**	**[H2]** A higher abundance of bioerosive (grazing and boring) species will be found on calcium rich rock (limestone) than lithologies without calcium (basalt and andesite)	(Coombes et al., 2011; Naylor et al., 2012b; Trudgill, 1988b)
Scottish sites selected from MarClim sites	**1**	**[H3]** Higher numbers and a greater variety of species will be found in microhabitats than on adjacent surfaces	(Firth et al., 2013; 2014b; Harper and Williams, 2001; Judge et al., 2009; Liversage et al., 2017; Menge et al., 1983; Strain et al., 2017b)
(Chapter 2) • Girvan • Dunure Castle	**2/3**	**[H4]** Pools will have greater species richness and abundance than other microhabitats.	(Aguilera et al., 2014; Firth et al., 2013; 2014a; 2014b)
• Eyemouth • Cove Harbour	**2/3**	**[H5]** A higher percentage of water held significantly increases species abundance and richness.	As above.
(2) Site scale (within 50 m) rock material and	**1**	**[H6]** The Barns Ness sandstone will be more ecologically suitable (lighter colour, more porous) than the limestone.	(Coombes, 2011; Coombes and Naylor, 2012; Sempere-Valverde et al., 2018)
rock mass study - Barns Ness, Scotland	**1**	**[H7]** The most ecologically suitable rock from laboratory testing will have greater species	(Coombes et al., 2011; Coombes and Naylor, 2012; Firth et al., 2012; McGreevy, 1985; Naylor

	(limestone and sandstone shore platforms)		richness and mobile species abundance in the field.	et al., 2012b; Sempere-Valverde et al., 2018)
		2	**[H8]** Higher numbers and a greater variety of species will be found in microhabitats than on adjacent surfaces	(Firth et al., 2013; 2014b; Harper and Williams, 2001; Liversage et al., 2017; Menge et al., 1983; Strain et al., 2017b)
		2/3	**[H9]** Pools and crevices (more abundant on the sandstone from ground truthing) will have greater species richness and abundance than other microhabitats.	(Aguilera et al., 2014; Bergeron and Bourget, 1986; Evans et al., 2015; Firth et al., 2014a; Jackson, 2010; Judge et al., 2009; Raffaelli and Hughes, 1978)
(3)	Shore platform scale - sandstone platform integrating dGPS with ecological survey data	**2/3**	**[H10]** A higher percentage of water held significantly increases species abundance and richness.	(Aguilera et al., 2014; Firth et al., 2013; 2014a; 2014b)
(4)	Shore platform scale- Glamorgan, South Wales	**2**	**[H11]** Density of jointing can differ within the same bed layer.	(Stephenson and Naylor, 2011)- shows high variability in discontinuities between adjacent layers at the same site.
		2	**[H12]** A more densely jointed area will have greater numbers of crevices and these microhabitats will increase richness and abundance of species compared with a less densely jointed area.	(Aguilera et al., 2014; Moreira et al., 2007; Stephenson and Naylor, 2011)
		2/3	**[H13]** Crevices and pools have higher species richness and mobile species abundance than adjacent areas on the platform surface.	(Aguilera et al., 2014; Bergeron and Bourget, 1986; Evans et al., 2015; Firth et al., 2014a; Jackson, 2010; Judge et al., 2009; Raffaelli and Hughes, 1978)

3.3 Methods

3.3.1 Common methodology

For the ecological sampling in Study 1,2 and 4 (Table 3-2), the ecological sampling methods were kept consistent. Sampling was conducted within a 10x10 m area, with three transects placed perpendicular to the shore at similar distances apart within the mid-upper intertidal zone, between Mean Tide Level (MTL) and Mean High Water Springs (MHWS), at each site. Ten quadrats (25x25 cm) were randomly placed along the right-hand side of every ten metre transect (n=3 transects, n=30 quadrats per site) using a random number generator to assign their location along the transect, with a minimum distance of 25 cm between adjacent quadrats. For Study 3, an area of 10x6 m was delineated on the sandstone platform at Barns Ness with belt transects used to capture ecology within the entire survey area. Methods for study 3 are explained in depth in Section 3.3.6.

To account for variation in species distributions between microhabitats, when quadrats fell on microhabitats the number, type and % cover of features within quadrats were recorded alongside the width, depth and water holding % of individual microhabitats. The number and type of each microhabitat was recorded alongside their water holding capacity. Depth of pools was taken from their deepest point and depth of cracks and crevices was taken as sidewall height to the nearest mm. Sidewall height (length from bottom to top of feature, with tape measure kept as close to the sidewall as possible) was also used to measure overhang depth (Bergeron and Bourget, 1986). Width was measured as the distance between two opposing walls for cracks and crevices (Bergeron and Bourget, 1986; Richter et al., 2001). Quadrats were placed flat on rock and species counted within this area and within the microhabitats within the quadrat boundaries.

To better understand the importance of microhabitats created through rocky shore processes such as pits, cracks, crevices, ledges etc., the presence and location of macrofauna and macroflora was recorded for each individual microhabitat and the platform surface that fell within quadrat boundaries. This was an extensive and time-consuming process due to the high spatial heterogeneity and diversity of habitats available on rocky shores (Gonor and Kemp, 1978). The area under algal holdfasts was also examined but not removed. Sampling included data both from the platform surface and the microhabitats present as it was important to understand how microhabitats played a role in the distribution of species. This has been previously stated to reduce the definition of data, i.e. the ability to sample every habitat to a high standard, required for either habitat (Gonor and Kemp, 1978) but the extensive nature of these surveys reduces this risk.

All macrofauna and macroflora taxa in each quadrat were visually recorded using non-destructive techniques and identified to species level where possible and where not, were identified to genus level such as coralline crusts 'Lithothamnion'. Counts were performed for mobile species while most sessile organisms (i.e. sessile fauna (barnacles), algae and lichen) were recorded as percentage cover. For mussels, anemones and *Polydora ciliata,* counts were performed. These species were classified as 'sessile fauna counts' as this provided more of a representation of their abundance. Percentage cover of barnacles were recorded and grouped for multiple species to speed up data collection, species identification at each site was confirmed later using photos collected during sampling. Where macroalgae or barnacles were present but had extremely limited cover, an arbitrary value of 0.5% was allocated for analysis.

3.3.2 Geomorphic features, rock material and rock mass properties – Comparing Scottish sites on the east and west coast

Four natural rocky shores were surveyed using comparable methodologies, with details of each site listed in Table 3-3 and associated photos in Figure 3-4. Sites were selected from the MarClim dataset that was previously examined for a coarse scale analysis (Chapter 2) to identify whether the influence of lithology was more identifiable at the site and regional scale (10s of km within regions, 100s of km between east and west coast of Scotland). The specific sites selected from the MarClim dataset for further field study were chosen due to their proximity and in their provision of a contrast of sedimentary and igneous lithologies on each coast as well as providing a high and low calcium contrast on the west coast. Each site was surveyed over a single low tide period in August 2018.

Table 3-3. Natural shore locations and lithology.

Site	Coast (Scotland)	Location	Lithology
Dunure Castle	West	55° 24' 20.8116'' N 4° 45' 50.3964'' W	Basalt and Basaltic Andesite
Girvan	West	55° 12' 34.5492'' N 4° 53' 32.4744'' W	Limestone and [Subequal/Subordinate] Argillaceous Rocks Interbedded
Eyemouth	East	55° 52' 29.9244'' N 2° 5' 31.956'' W	Andesite and Basalt
Cove Harbour	East	55° 56' 20.634'' N 2° 20' 41.9676'' W	Sandstone

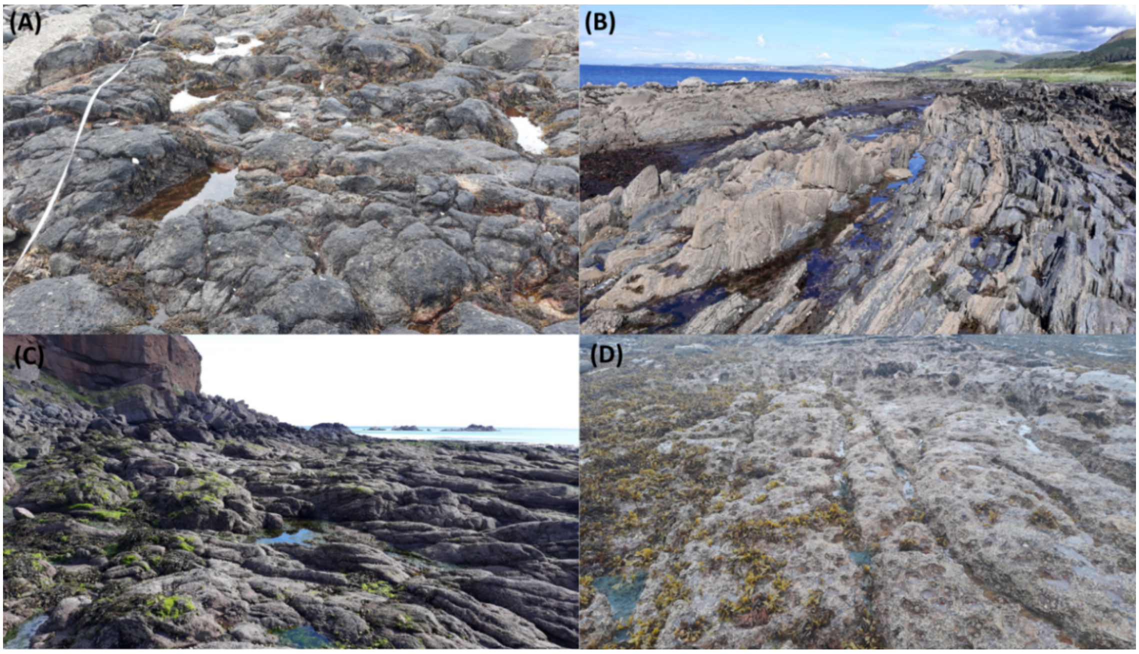

Figure 3-4. Natural rocky shore survey sites (A) Dunure Castle (B) Girvan (C) Eyemouth (D) Cove Harbour.

3.3.3 Barns Ness field studies

Study 2 compared two different lithologies within the same exposure and environmental conditions. These were conducted on a sandstone and adjacent limestone rocky shore platform less than 50 metres away at Barns Ness, East Lothian (55° 59' 16.35" N, 2° 27' 33.78" W) in July 2018. Results from Study 2 were combined with background information gathered from laboratory tests on the two lithologies to better understand how rock material and rock mass properties influence geomorphic features present and their combined effects on the spatial distribution, richness and abundance of intertidal ecology. Both platforms were gently sloping, relatively smooth surfaces interspersed with a variety of microhabitats including crevices and pools. This was determined from geological maps and subsequent ground truthing.

Study 3 surveyed the same sandstone platform as study 2 in August 2017, combining ecological surveys with terrestrial laser scanning (TLS) and DGPS to create a density map of the survey area. Only the sandstone platform was surveyed as this platform was found to have more jointing, with the limestone platform later shown to be modified by abrasion. Focusing the detailed surveying on the platform with more microhabitats aided in further informing Research Question 2.

3.3.4 Barns Ness- rock material tests

Albedo, porosity, water absorption capacity and surface roughness, which are known to influence ecological suitability, were tested using samples of the limestone and sandstone platforms to determine if rock material property potentially influenced species distributions. These laboratory tests were undertaken by a University of Glasgow undergraduate student (Senogles, 2017), with all statistical analyses and results conducted personally as part of this thesis. Measurements were made using rough (broken), smooth (cut) and weathered samples.

For measurements of surface roughness and albedo, samples were cut into 6 cm^2 sections with n=3 rough (broken by hammer), n=3 smooth (saw cut) and n=3 weathered samples of each lithology. Samples were placed next to a control surface of white paper (albedo value of 0.65 (Gorski, 2011) and three photographs were taken of samples at 9am, 12pm, 4pm and under lab light conditions, increasing accuracy by measuring albedo under different light conditions. Photos were then uploaded to ImageJ and a histogram of reflective light was obtained and the mean brightness value was recorded. Albedo was calculated using equation 1*,* where Bs is mean sample brightness, Bp is mean paper brightness and Ap is the albedo paper value (Gorski, 2011). Values were calculated for dry samples as the low tide period is the most stressful for intertidal species.

Equation 1: *Albedo (A) = (Bs/Bp) x Ap*

One side of each of the 6 cm^2 samples from both lithologies was scanned using a NextEngine laser scanner at 100µm resolution at 8 cm distance. The resulting point clouds created were imported into ArcMap and converted using an inverse distance weighted (IDW) technique to create a DEM surface, with the area clipped to the scan surface. Surface roughness was quantified using standard deviation of slope after Grohmann *et al.* (2010), which was considered the most suitable method due to its accuracy in analysing the whole surface area and ability to examine roughness across multiple scales. Pixel neighbourhoods were created around each pixel in the DEM to calculate standard deviation of elevations within this neighbourhood at the 0.25 mm^2 and 25 mm^2 scales, scales considered appropriate for the small dimensions of samples (maximum 6 cm^2).

For water absorption and porosity tests, seven 9 cm^3 cubes of each lithology were cut from the centre of each rock to ensure unweathered samples. Cubes were kept in distilled water overnight and dried the next day in order to remove any potential sediment remaining in pore spaces. All but one side of the cubes was coated in two thin coats of marine varnish to allow for accurate measurements of porosity and WAC. Cubes were placed in a rock oven at 105°C for 24 hours to remove surplus moisture and subsequently placed in desiccators to cool to room temperature (~21°C) over two hours. Once cooled, the dry mass (Ms) and volume (V) of each cube was measured. The varnished cubes were submerged in seawater for seven days, allowing them to absorb water. After this submergence period the samples were dried of excess water and weighed to find saturated mass (Msat). Effective porosity was measured using equation 2 for pore volume and equation 3 for porosity calculations. WAC was calculated using equation 4 (Coombes, 2011).

Equation 2: *Pore Volume (Vv, cm³) =(Msat-Ms)*

$$\text{Pore Volume } (Vv, cm^3) = (Msat - Ms)$$

Equation 3: *Porosity (n, %) = ((Vv)/V) x 100*

Equation 4*: WAC (%)= (Msat-Ms) x 100*

3.3.5 Barns Ness- terrestrial laser scanning of the platforms

Terrestrial laser scans (TLS) were made using a TLS Leica C10 to gather surface elevation profile data and capture an overview of the rocky shore platforms. TLS data was collected during extremely low spring tides allowing complete coverage of the mid-upper level of the intertidal platform. Although the platforms were relatively smooth, multiple scan stations were required to attempt to capture the complexity of the shore. The scanner was moved to 4 locations on the sandstone platform and 6 on the limestone platform to minimise the effects of shadowing and to capture data from different angles. Scans of the limestone platform were used to compare surface profiles but were not used in ecological analyses. Scans were taken at a resolution of 1 mm and a distance ranging from 5-15 metres, with the sandstone platform measuring approximately 257.5 m^2 and the limestone measuring 102.4 m^2 in area. Circular targets were placed to function as reference points for scans and aid in overlapping scans when processing the data. Scans were processed in Leica Cyclone 9.0, with the point clouds for each platform cleaned, merged and exported to ArcMap to create a DEM and a mesh created of the surface to better highlight geomorphic features on the platforms. Profile graphs were created using the 3D Analyst toolbar to highlight the variation of profiles on the platforms.

3.3.6 Barns Ness- density map creation on the sandstone platform

A 10x6 metre area of the sandstone platform ranging in height from 2.05-2.23 m above sea level, within the area where the TLS were conducted, was delineated using multiple transects. A total of 240 quadrats (50 cm^2) were sequentially placed in this area and differential GPS (DGPS) was used to record the latitude and longitude location of each quadrat (Figure 3-5). Species were recorded in each quadrat using the standard sampling methodology (3.3.1) and the percentage of water held in each quadrat (inside features) was recorded alongside the total number of features. Points (DGPS) were imported into ArcMap with associated quadrat data and overlaid onto the DEM of the platform surface. Raster surfaces were interpolated using an inverse distance weighted technique to create a DEM, with the area clipped to the extent of the sampling area. The 'height' of the DEM was representative of the numerical values for species richness, mobile abundance, total features and water holding (%) per quadrat (50 cm^2), resulting in a hotspot density map of the platform that automatically in-filled the few spots where quadrats were not perfectly aligned.

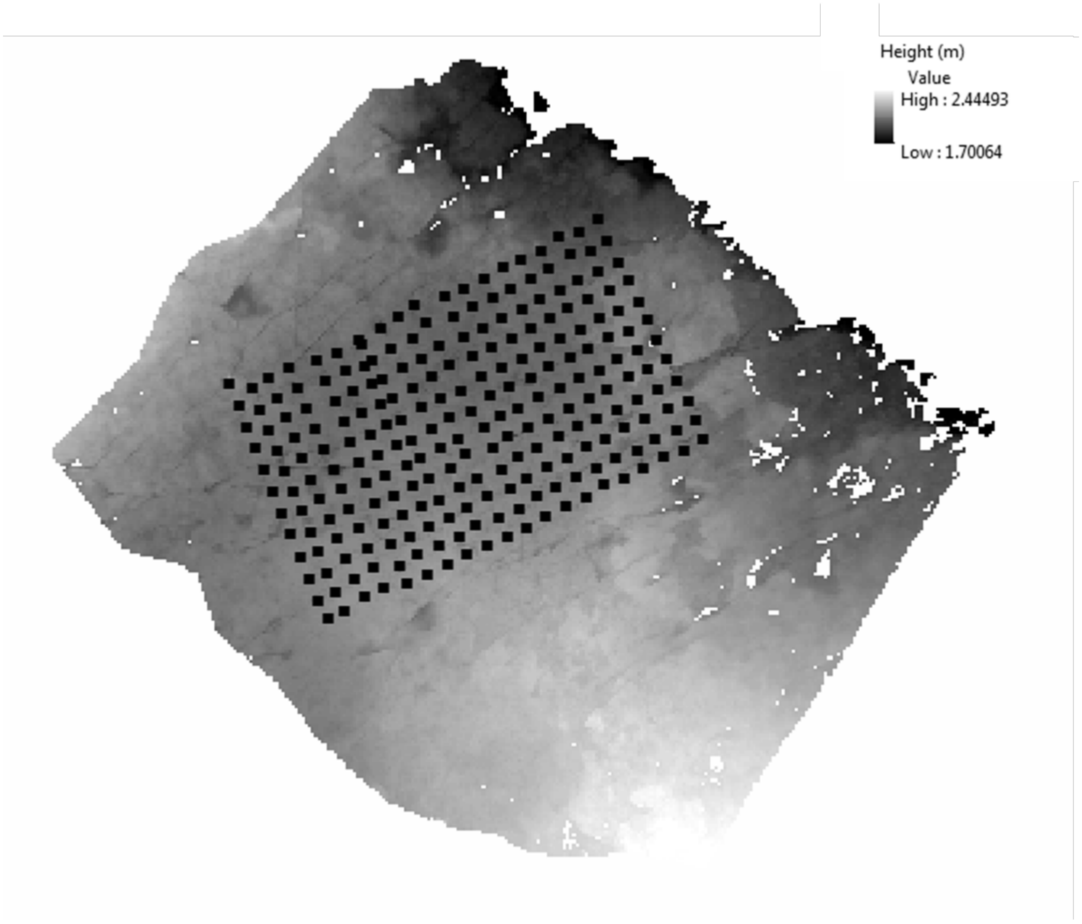

Figure 3-5. DGPS points in black (n=240) overlaid onto the DEM of the platform surface with height in metres above sea level.

3.3.7 Glamorgan coast- joint density within the same bed layer

Data were collected on two adjacent areas of a section of an exposed Blue Lias limestone rocky shore platform on the Glamorgan coast, Wales (51°23’48.357” N, 3°32’25.6446”W) (Figure 3-6), situated within the Bristol Channel. The Blue Lias limestone platform slopes seawards (3° dip, 185° dip direction), with platform widths typically in the range of 200-250 m (May and Hansom, 2003; Naylor et al., 2016). The two areas of the shore platform were selected for ecological sampling so that there was no variation in background environmental conditions (e.g. wave exposure, temperature) and lithology (i.e. they were from the same bed layer), allowing a clear comparison of the influence of microhabitat type and density on biodiversity metrics. It has been noted that within sections of this platform, there is large variability in discontinuities and the presence of microhabitats between and within individual layers (Stephenson and Naylor, 2011). The typical tidal range at this site is 6 m, with ecological quadrat surveys conducted over a single low-tide in November 2017.

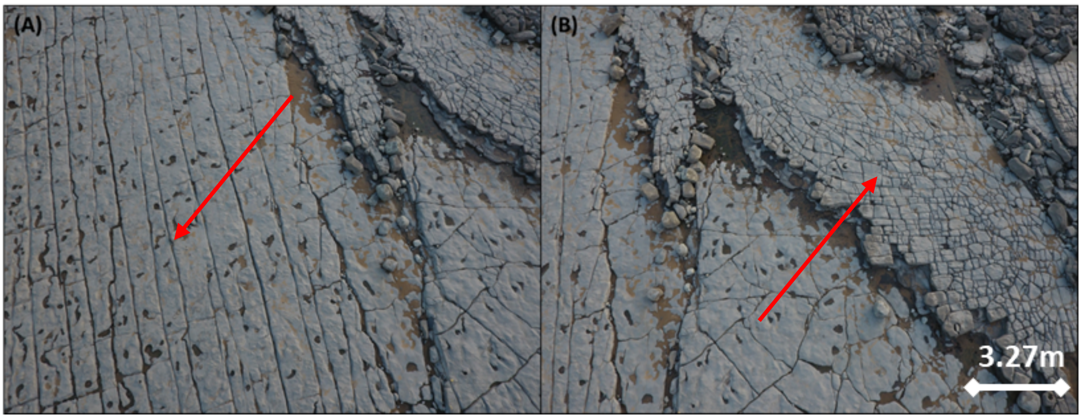

Figure 3-6. (A) Less densely jointed and (B) more densely jointed sections of the intertidal shore platform at Glamorgan. Red arrow highlights area of survey.

Surveys were conducted within the same bed layer but in two separate sections, one that was sparsely and the other densely jointed to allow comparison and answer Hypothesis 12 (Table 3-2). At this site, five microhabitats were present in addition to the platform surface: (1) hairline cracks (2) cracks (3) crevices (4) small ledges and (5) shallow intertidal rock pools.

3.4 Statistical Analyses - Scottish sites (east and west coast)

All analyses in this chapter were carried out in R version 3.5.1 (R Development Core Team 2018). The lithologies on each coast were conflated (i.e. collapsed into sedimentary and igneous lithologies) as both coasts had different lithologies but it was expected that sedimentary lithologies would perform better in the examined ecological variables. Negative binomial, Quasi-Poisson and Binomial Generalised Linear Models (GLM) were then used to test whether species richness and abundance were influenced by lithology, coast and the interaction between lithology and coast. These methods were selected as they resolve issues with the high volume of zero observations and subsequent over-dispersion of count data. Residuals and the Akaike Information Criterion (AIC) (where available) were examined to determine the most appropriate model to use (Zuur et al., 2009). For non-integer data (algae and lichen abundance (%) and barnacle abundance (%)), data were converted to logits (ln(P/(100-P)), with a trap for 100% and 0% (replaced with 0.5% and 99.5%) and analysed as normal with a Gaussian link function. Where algae abundance exceeded 100%, this data was excluded from analyses (n=3 quadrats).

Following this, statistical differences in species richness, abundance, diversity and feature abundance (crack, crevice, pool etc.) were tested on each coast with the fixed factor 'lithology'. A combination of Negative Binomial and Quasi-Poisson GLMs were used (Table 3-4). For non-integer data (algae and lichen abundance (%), barnacle abundance (%), diversity and water holding features (%)), Kruskal-Wallis tests were used to test the interaction between these variables and lithologies at each coast.

Table 3-4. Models selected to analyse variables for each coast.

	East Coast	**West Coast**
Species richness	GLM- Quasi-Poisson	GLM- Quasi-Poisson
Mobile abundance	GLM- Negative binomial	GLM- Negative binomial
Sessile fauna count (*Polydora* sp., Mussels, Anemones)	N/A	GLM- Negative binomial
Species Diversity	Kruskal-Wallis	Kruskal-Wallis
Algae and lichen abundance	Kruskal-Wallis	Kruskal-Wallis
Barnacle abundance	Kruskal-Wallis	Kruskal-Wallis
	All lithologies	
Total number of features	GLM- Negative binomial	

The first hypothesis (Table 3-2), that softer lithologies (limestone and sandstone) will have greater species richness, abundance (mobile and sessile species) and diversity than harder lithologies (basalt and andesite), was tested. Shannon Wiener diversity index (H) was calculated for each quadrat within each lithology. Coasts were analysed separately for all ecological variables. Sessile fauna count data was excluded from analyses on the east coast due to the extremely limited presence of these species. Hypothesis 2 was that a greater number of bioerosive species (grazing and boring species) would be found on the calcium rich lithology (limestone) than lithologies lacking calcium. This focused on *Patella vulgata*, a known bioeroder, and *Polydora ciliata,* the only boring species recorded. Bioprotective species were also identified, i.e. barnacles and macroalgae. Negative binomial GLMs were conducted on limpet abundance (known bioeroder) and Kruskal-Wallis tests for barnacle and algae and lichen abundance in Hypothesis 1 were used. *Polydora ciliata* could not be statistically analysed as it was only present on the limestone shore at Girvan. Patterns in the abundance of these species were then outlined to answer the hypothesis.

Hypothesis 3 and 4 examine the importance of microhabitats, with higher richness and abundance of species anticipated in these geomorphic features than on adjacent surfaces and in pools compared to other habitats. Comparisons with microhabitats (Negative Binomial GLM) and water holding capacity (Kruskal-Wallis) did not differentiate between coasts as these analyses were used solely to determine if different lithologies had different rock mass properties. A new dataset was created to examine microhabitats to determine whether they exerted an influence on ecological variables. Quadrats were classed into those with and without each microhabitat, with every recorded feature inputted as an individual row in the dataset so that ecological variables could be attributed to the feature. For each quadrat where features did not cover 100% of the area, the platform surface and its associated ecological variables were recorded. GLMs were used to test for differences between feature types (e.g. cracks, crevice, deep pool, ledge, surface) for species richness and mobile species abundance within each lithology. Kruskal-Wallis tests were used to test differences in algae and lichen abundance (%), barnacle abundance (%) and sessile fauna counts between features. Generalised linear hypothesis testing (GLHT) with Tukey's pairwise comparisons used for all GLM post-hoc comparisons and Dunn's post hoc tests with Bonferroni corrections used for multiple comparisons with Kruskal-Wallis. Finally, hypothesis 5, that the presence of water holding features significantly increases species abundance and richness, was evaluated using Pearson's correlation coefficient, to determine if there was a linear relationship between water holding % and ecological variables for each lithology.

3.5 Statistical Analyses- Barns Ness

For laboratory comparisons between rock material properties, Kruskal-Wallis tests were used with Dunn's post hoc tests where multiple comparisons were required. For all field comparisons, GLMs (Quasi-Poisson or Negative Binomial) were used to resolve issues with over-dispersion of count data as mentioned above. Kruskal-Wallis tests were used for algae and lichen abundance (%). These tests allowed for comparisons between the two different lithologies. Additionally, as above, microhabitat data was extracted and the same method of analysing the data was applied, with GLHT Tukey's comparisons and Kruskal-Wallis used for comparing multiple features within each platform.

For Study 3 (Table 3-2), Pearson's product-moment correlation was conducted between species richness, mobile species abundance, algae and lichen abundance (%) and water holding (%) in quadrats (50 cm^2) and total features on the sandstone platform at Barns Ness (n=240 quadrats).

3.6 Statistical Analyses- Glamorgan

To examine whether there was a difference in species richness and mobile species abundance recorded in densely versus sparsely jointed survey areas, Negative Binomial or Quasi-Poisson GLMs were applied to the data. These models were used due to the high number of zero observations in the count data and resulting over-dispersion. The most suitable model was selected after examination of the residuals and the resulting model with the lowest Akaike Information Criterion (AIC) was selected. In later analyses of optimal habitat type within each survey area, a new dataset was created using the same methods in Section 3.4. This involved analysis of microhabitat types, with every microhabitat found in quadrats a separate row in the dataset. Where microhabitats did not cover 100% of the platform surface, "Surface" was counted as a row for analysis (n=60, n=30 per survey area). This dataset only included features where they were present for a direct comparison of habitat suitability, with the original dataset rearranged so that features were the main focal point of the secondary analyses. GLMs were used to test for differences between microhabitats for species richness and mobile species abundance within each jointing density. Generalised linear hypothesis testing (GLHT) with Tukey's pairwise comparisons were used for all GLM post-hoc comparisons.

3.7 Results - Scottish sites (east and west coast)

Adding in the interaction between coast and lithology highlights that for mobile species abundance and sessile fauna counts, the contrast between limestone and basalt produced most of the significant differences. Limestone has significantly greater mobile species abundance and sessile fauna species abundance than all other examined lithologies (Table 3-5). For overall species richness and algae and lichen abundance the effects of both coast and lithology did not exert any influence. Barnacle abundance was found to be significantly lower on the west coast than the east coast ($p<0.001$, Table 3-5).

Table 3-5. GLMs for a) species richness, b) mobile species abundance, c) sessile fauna abundance (counts), d) algae and lichen abundance, e) barnacle abundance in relation to lithology merged (sedimentary and igneous) and coast (west and east). Terms are condensed into lithology merged (Lmerge), the associated lithology (sedimentary, igneous) and then the coast (***= p<0.001, **=p<0.01, *=p<0.05, NS= Not significant).

a) Species richness (Quasi-Poisson)				
	Estimate	**Std. Error**	**T value**	**P-value**
(Intercept)	1.224	0.093	13.213	***
LmergeSedimentary	0.187	0.125	1.495	NS
Coast West	0.029	0.130	0.223	NS
LmergeSedimentary:Coast West	0.329	0.170	1.934	NS
b) Mobile species abundance (Negative Binomial) theta = 1.0855				
	Estimate	**Std. Error**	**Z value**	**P-value**
(Intercept)	2.536	0.183	13.889	***
LmergeSedimentary	-0.065	0.259	-0.253	NS
Coast West	-0.365	0.260	-1.403	NS
LmergeSedimentary:Coast West	1.085	0.365	2.970	**
c) Sessile fauna abundance (counts) (Negative Binomial) theta= = 0.2708				
	Estimate	**Std. Error**	**Z value**	**P-value**
(Intercept)	-1.609	0.538	-2.990	**
LmergeSedimentary	-1.099	0.955	-1.150	NS
Coast West	2.197	0.657	3.346	***
LmergeSedimentary:Coast West	3.560	1.086	3.279	**
d) Algae and lichen abundance (Gaussian)				
	Estimate	**Std. Error**	**T value**	**P-value**
(Intercept)	-2.472	0.419	-5.897	***
LmergeSedimentary	-0.074	0.588	-0.125	NS
Coast West	-0.771	0.583	-1.323	NS
LmergeSedimentary:Coast West	1.145	0.821	1.395	NS
e) Barnacle abundance (Gaussian)				
	Estimate	**Std. Error**	**T value**	**P-value**
(Intercept)	-1.809	0.342	-5.286	***
LmergeSedimentary	-0.210	0.484	-0.434	NS
Coast West	-1.693	0.484	-3.497	***
LmergeSedimentary:Coast West	1.119	0.685	1.634	NS

Limestone performed statistically better than basalt across all examined ecological variables of richness, abundance and diversity (Table 3-6, Table 3-7). On the east coast, sandstone was significantly more diverse than andesite ($H(1)=4.133$, $p<0.05$) and had a higher species richness, although this was not statistically significant (Figure 3-7). When comparing between coasts, limestone had higher species richness, diversity and abundance on average than other lithologies across both coasts, with the exception of algae and lichen and barnacle abundance, which was highest on andesite (Figure 3-7, Figure 3-8). Sessile fauna abundance could not be analysed for the east coast as very few of these species were recorded resulting in a lack of data, with a total of four *Actinia equina* and two *Mytilus edulis* over two quadrats (out of a total of n=30) on the andesite and two *Mytilus edulis* in one quadrat on the sandstone (n=30).

Table 3-6. Summary of GLM results (Negative Binomial and Quasi-Poisson) for species richness, mobile abundance and sessile fauna abundance (counts of *Mytilus edulis, Actinia equina* and *Polydora ciliata*) between lithologies on the A) east coast and B) west coast of Scotland (***=p<0.001, **=p<0.01, *=p<0.05, NS= Not significant).

A) West coast	**Species richness Quasi-Poisson**				**Mobile abundance Negative Binomial**			
	Estimate	**Std. Error**	**T value**	**P**	**Estimate**	**Std. Error**	**Z value**	**P**
Intercept (Basalt)	1.253	0.098	12.801	***	2.171	0.193	11.224	***
Limestone	0.517	0.124	4.177	***	1.020	0.269	3.789	***
	Sessile fauna abundance (count) Negative Binomial							
	Estimate	**Std. Error**	**Z value**	**P**				
Intercept (Basalt)	0.5878	0.3538	1.661	NS				
Limestone	2.4615	0.4831	5.095	***				
B) East coast	**Species richness Quasi-Poisson**				**Mobile abundance Negative Binomial**			
	Estimate	**Std. Error**	**Z value**	**P**	**Estimate**	**Std. Error**	**Z value**	**P**
Intercept (Andesite)	1.224	0.085	14.327	***	2.536	0.175	14.497	***
Sandstone	0.187	0.116	1.621	NS	-0.065	0.248	-0.264	NS

Table 3-7. Kruskal-Wallis results for algae and lichen abundance, barnacle abundance and species diversity between lithologies on the A) west coast and B) east coast of Scotland (***=p<0.001, **=p<0.01, *=p<0.05, NS= Not significant).

a) West coast	**Species diversity**			**Algae and lichen abundance (%)**			**Barnacle abundance (%)**		
Factor	**df**	***H*** **(X^2)**	**P**	**df**	***H*** **(X^2)**	**P**	**df**	***H*** **(X^2)**	**P**
Limestone- Basalt	1	11.783	*** L>B	1	4.240	* L>B	1	4.406	* L>B
b) East coast	**Species diversity**			**Algae and lichen abundance (%)**			**Barnacle abundance (%)**		
Factor	**df**	***H*** **(X^2)**	**P**	**df**	***H*** **(X^2)**	**P**	**df**	***H*** **(X^2)**	**P**
Sandstone-Andesite	1	4.133	* S>A	1	0.011	NS	1	0.552	NS

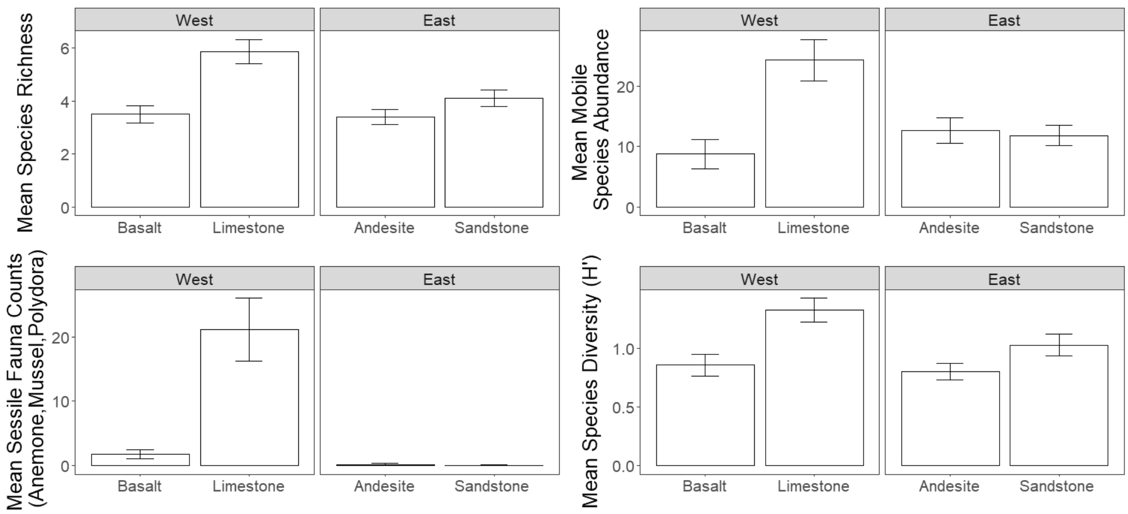

Figure 3-7. Mean species richness, mobile species abundance, sessile fauna counts (Anemones, Mussels, *Polydora ciliata*) and diversity for lithologies on the east and west coast of Scotland (per 25 cm^2, n=30 quadrats per site) ($\bar{x}$ ± standard error).

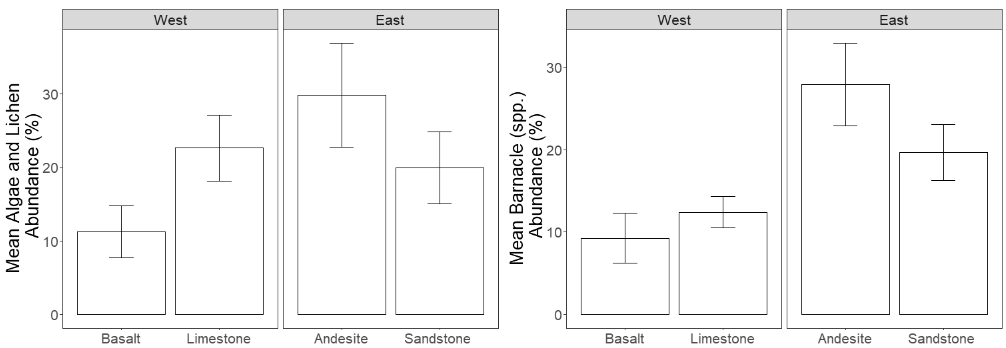

Figure 3-8. Mean algae and lichen abundance (%) and barnacle abundance (%) (per 25 cm^2, n=30 quadrats per site) ($\bar{x}$ ± standard error).

A total of 29 species were recorded across the four sites, with 21 species recorded at each coast. The number of species was very consistent between sites, with all except the sandstone (n=18) having 16 species. Five species, predominantly mobile fauna, were unique to limestone surveys (*Littorina obtusata, Polydora ciliata, Bdellidae* sp.*, Carcinus maenas* and *Nucella lapillus*), which was equally matched by five unique species on sandstone that largely consisted of algae (*Fucus vesciulosus, Anurida maritima, Mastocarpus stellatus, Porphyra umbilicalis* and *Verrucaria* sp.) (Table 3-8). Only three species (*Littorina saxatilis, Rhodothamniella floridula, Lichina pygmaea*) were unique to basalt and one to andesite (*Fucus serratus*). Limestone supported a greater abundance of mobile species, particularly gastropods, and heavily calcified species (*Corallina officialis* and *Lithothamnion* sp.) than the other examined lithologies (Figure 3-9). Barnacles are a bioprotective species, i.e. a species that actively or passively protects the substratum from weathering processes (Coombes et al., 2013b). Barnacles were most abundant on andesite, with the second highest abundance recorded on sandstone (Figure 3-9, Table 3-8). *Semibalanus balanoides* was ubiquitous at the examined sites, with *Chthamalus montagui* also noted at both west coast sites.

In examining the types of species in terms of their biogeomorphological contributions, higher abundance of grazing and boring species (gastropods and *Polydora ciliata* respectively) was found on the softer limestone than harder lithologies. Limpet abundance was lower on the west coast than east coast ($z=-4.157$, $p<0.001$) but there were no differences between lithologies on either coast (west: $z= 0.806$, $p=0.420$, east: $z=-0.753$, $p=0.451$). Bioprotective species varied in their distributions with seaweeds more abundant on andesite on the east coast (28.01% (sum of mean abundance) compared to 18.62% for sandstone, Table 3-8) and limestone on the west coast (20.42% compared to 10.32 % with basalt) and encrusting coralline species most common on andesite and limestone (Figure 3-9).

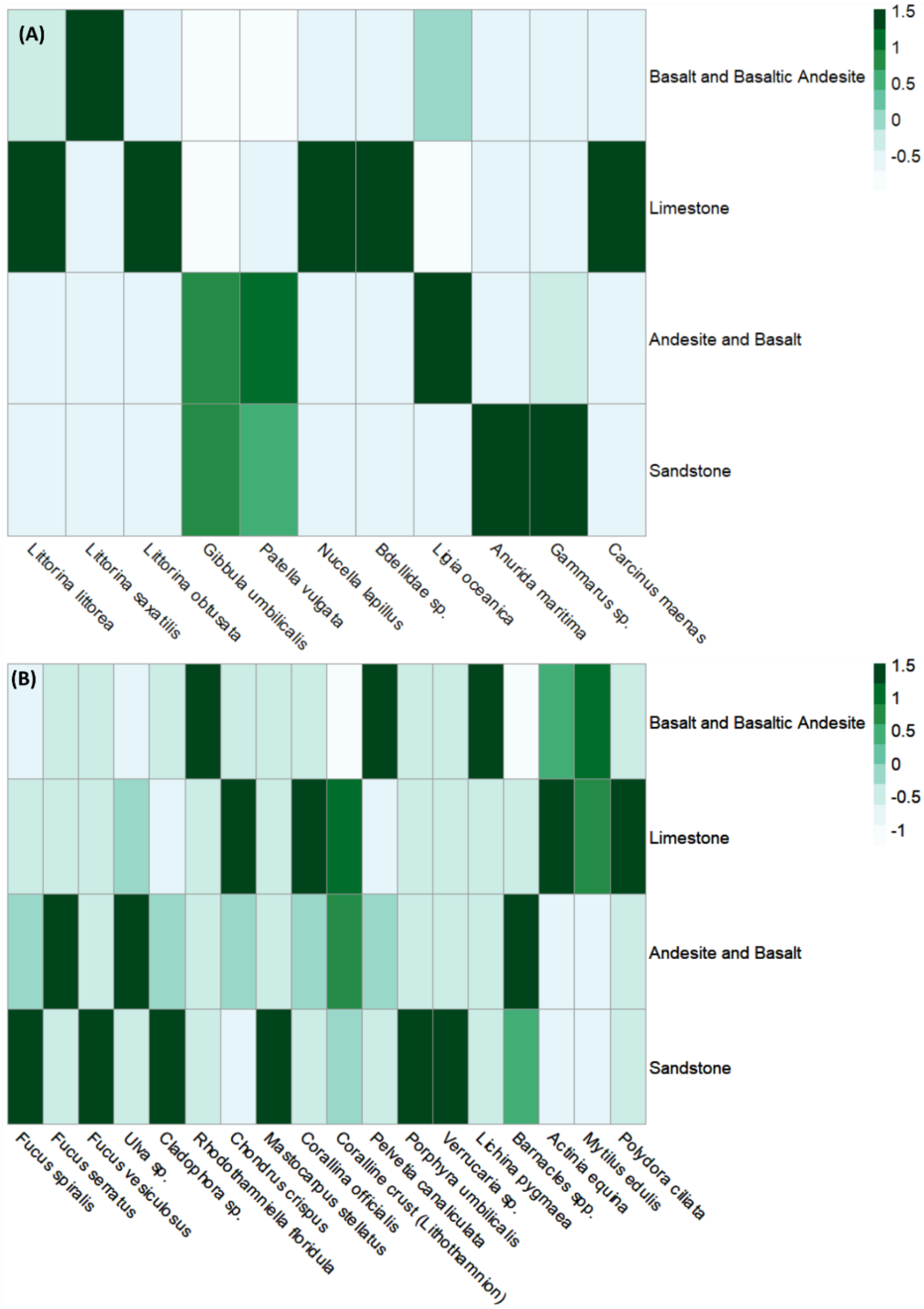

Figure 3-9. Scaled heatmap by mean species abundance for (A) mobile species (B) sessile algae and fauna with lithology. A value of 1-1.5 indicates high abundance and -0.5 or -1 indicates low abundance or absence (lowest value for each).

Table 3-8. Mean species abundance (± SE) per quadrat (25 cm^2) by lithology, c= counts and % = percentage cover. X indicates absence.

	West coast		**East coast**	
Species	Dunure Castle (Basalt and Basaltic Andesite)	Girvan (Limestone)	Eyemouth (Andesite and Basalt)	Cove Harbour (Sandstone)
***Littorina littorea* c**	4.80 (± 2.10)	19.30 (±3.48)	0.50 (±0.17)	1.53 (±0.39)
***Littorina saxatilis* c**	0.27 (±0.14)	X	X	X
***Littorina obtusata* c**	X	0.03 (±0.03)	X	X
***Gibbula umbilicalis* c**	X	X	0.03 (±0.03)	0.03 (±0.03)
***Patella vulgata* c**	3.57 (±0.80)	4.80 (±1.04)	11.50 (±2.08)	9.40 (±1.37)
***Nucella lapillus* c**	X	0.03 (±0.03)	X	X
***Carcinus maenas* c**	X	0.03 (±0.03)	X	X
***Actinia equina* c**	1.00 (±0.50)	1.87 (±0.41)	0.13(±0.10)	X
***Mytilus edulis* c**	0.80 (±0.41)	0.73 (±0.33)	0.07(±0.07)	0.07 (±0.07)
***Polydora ciliata* c**	X	18.50 (±4.87)	X	X
***Bdellidae sp.* c**	X	0.10 (±0.07)	X	X
***Ligia oceanica* c**	0.13 (±0.13)	X	0.47 (±0.33)	0.03 (±0.03)
***Anurida maritima* c**	X	X	X	0.17(±0.14)
***Gammarus sp.* c**	X	X	0.13(±0.13)	0.67(±0.51)
***Fucus spiralis* %**	1.37(±0.57)	1.83 (±0.66)	4.17 (±2.04)	12.45 (±3.55)
***Fucus serratus* %**	X	X	1.67 (±1.67)	X
***Fucus vesiculosus* %**	X	X	X	0.83 (±0.59)
***Ulva sp.* %**	0.53 (±0.35)	4.75 (±1.39)	17.67 (±5.44)	1.17 (±0.46)
***Rhodothamniella floridula* %**	0.42 (±0.34)	X	X	X
***Chondrus crispus* %**	0.50(±0.37)	5.42(±1.61)	1.17(±0.57)	X
***Mastocarpus stellatus* %**	X	X	X	1.67 (±0.81)
***Cladophora sp.* %**	0.75 (±0.32)	0.67 (±0.40)	1.00(±0.74)	1.67 (±0.81)
***Corallina officialis* %**	0.17(±0.17)	7.75(±1.95)	1.33(±0.68)	0.33(±0.33)
Coralline crust (Lithothamnion sp.) %	0.17(±0.17)	2.20(±1.39)	1.83(±1.41)	1.00(±0.56)
***Pelvetia canaliculata* %**	6.58(±2.86)	X	1.00(±1.00)	0.33(±0.33)
***Porphyra umbilicalis* %**	X	X	X	0.17(±0.17)
***Verrucaria sp.* %**	X	X	X	0.33(±0.33)
***Lichina pygmaea* %**	0.75(±0.47)	X	X	X
Barnacle spp. (*Semibalanus balanoides* or *Chthamalus montagui*) %	9.27 (±3.00)	12.42 (±1.89)	27.92 (±5.00)	19.67 (±3.40)
Total number of species	16	16	16	18

Different types and numbers of microhabitats were recorded at each site. Using a negative binomial GLM, basalt (z=3.580, p=0.002), limestone (z=2.970, p=0.016) and sandstone (z-3.694, p=0.001) had significantly more microhabitats than andesite. Basalt had the greatest diversity of microhabitats, having 7 of the 8 recorded microhabitats (Table 3-1), with cracks unique to basalt (Figure 3-10). Microhabitats could only be statistically compared where they were present at all sites. Pits were only present on sandstone and limestone, with sandstone more pitted ($\bar{x}$=1.73 (±0.57), Figure 3-10). Shallow and deep pools were more abundant on limestone than other lithologies (pools combined: $\bar{x}$=0.37 (0.09± SE)), although the abundance of pools with lithology was not statistically significant. Water holding habitat (% of quadrat holding water) did not statistically differ between lithologies (*H*(3)=4.896, p=0.180). Ledge habitat was significantly greater on basalt and limestone than sandstone (Quasi-Poisson model, z=2.897, p=0.019 and z=3.540, p=0.002 respectively).

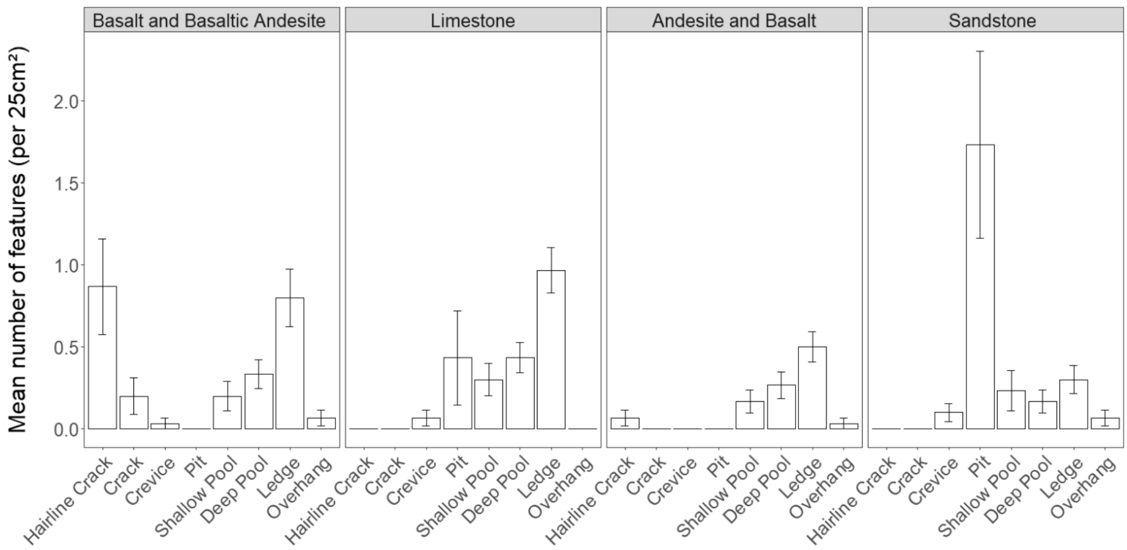

Figure 3-10. Mean number of features per quadrat (25 cm^2) for each lithology ($\bar{x}$ ± standard error).

In comparing ecological variables of richness and abundance with feature types, full tables of statistical results are available in Table A 3-1 and Table A 3-2. Across all examined lithologies, deep pool habitat had significantly greater species richness than the platform surfaces (p<0.05) (Table 3-9). Particularly on the limestone, species richness was greater in habitats with greater complexity such as shallow and deep pools, ledges and crevices than the platform surface (p<0.001), although deep pools were the optimal habitat as these also significantly exceeded the prevalent ledges in attracting higher numbers of species ($p<0.01$, Table 3-9). A similar pattern was observed with algae and lichen abundance, whereby deep pools exceeded ledge and marginal habitats (ledge adjacent (on the platform surface at the bottom of ledges) and pool adjacent (halo of wetted surface around the pool) on several lithologies (Table 3-9). Both deep and shallow pools were optimal for algae and lichen abundance over the aforementioned marginal habitat types on the limestone (Table 3-9). There was variation in which habitat was optimal for mobile species abundance, with deep pools hosting greater abundance than the platform surface and ledge adjacent habitats on both basalt and limestone ($p<0.05$). Overhang habitat was more effective at attracting mobile species than the platform surface and hairline cracks on the basalt, despite there being few overhangs recorded ($p<0.05$, Table 3-9, Figure 3-10). Ledge habitat was notably the most effective microhabitat on andesite for mobile fauna abundance, exceeding the platform surface and both shallow and deep pool habitats ($p<0.05$, Table 3-9). Pool and ledge habitat on the sandstone made very little difference to mobile fauna abundance, whereas both pits, which were found in high volume across the sampling area (Figure 3-10), and crevices were more suitable than the platform surface ($p<0.05$).

For sessile fauna abundance (*Actinia equina, Polydora ciliata* and *Mytilus edulis*), very few were recorded on andesite, with n=1 *Actinia equina* in a deep pool and n=3 *Actinia equina* and n=2 *Mytilus edulis* under an overhang. The same was found with sandstone, where n=2 *Mytilus edulis* were found in a crevice. As such, no statistical comparison could be made as these species were too sparsely recorded (Table 3-8). For basalt, feature type made little difference for the *Actinia equina* (1 ± 0.50 SE per quadrat) and *Mytilus edulis* (0.80 ± 0.41 SE) recorded, although cracks and hairline cracks had the highest numbers of mussels recorded (n=11 in a single crack and n=5 in a single hairline crack) while deep pools were the most favoured habitat of *Actinia equina* (n=24 anemones over 4 deep pools). On the limestone, regardless of depth, pool habitat offered the best refugia for sessile fauna, exceeding ledges and pits ($p<0.01$, Table 3-9).

Table 3-9. Significant post-hoc results for comparisons of species richness (Quasi-Poisson GLM), mobile abundance (Negative Binomial GLM), sessile fauna abundance (Negative Binomial) and algae and lichen abundance (Kruskal-Wallis tests) with feature types on each lithology. Feature abbreviations are as follows: S= surface, HC= hairline crack, Cre= crevice, SP = shallow pool, DP = deep pool, L = ledge, L adj= ledge adjacent, P adj = pool adjacent, O= overhang. NS = No significant results, N/A = too few data to allow for statistical testing (***=p<0.001, **=p<0.01, *=p<0.05).

	Basalt		**Limestone**		**Andesite**		**Sandstone**	
Species Richness (Quasi-Poisson)	DP > HC	***	Cre > S	***	DP > L	**	Cre > L adj	*
	DP > L	**	DP > L	**	DP > L adj	***	DP > L adj	***
	DP > L adj	***	DP > L adj	***	DP > P adj	***	DP > P adj	***
	DP > P adj	***	DP > P adj	***	DP > S	*	DP > S	*
	DP > S	***	DP > S	***	L > L adj	**	Pit > L adj	*
	O > L adj	*	L > L adj	***	O > L adj	**	SP > L adj	**
			L > P adj	***	SP > L adj	**	S > L adj	*
			L > S	***	S > L adj	***		
			SP > L adj	***				
			SP > P adj	***				
			SP > S	***				
Mobile abundance (Negative Binomial)	DP > HC	**	DP > L adj	***	L > DP	*	Cre > S	*
	DP > L adj	*	DP > P adj	**	L > L adj	***	Pit > L adj	*
	DP > S	***	DP > S	***	L > SP	*	Pit > S	***
	O > HC	*	L > L adj	***	L > S	**		
	L> S	*	L > S	*	S > L adj	**		
	O > S	*	Pit > L adj	*				
			SP > L adj	**				
Sessile fauna abundance (Negative Binomial)	NS		DP > L	***	N/A		N/A	
			DP > L adj	***				
			DP > Pit	**				
			DP > P adj	***				
			SP > L	**				
			SP > L adj	**				
			SP > P adj	***				
Algae and lichen abundance (Kruskal-Wallis)	DP > L	***	DP > L	***	DP > L	***	NS	
	DP > L adj	***	DP > L adj	***	DP > L adj	***		
	DP > S	***	DP > P adj	***	DP > P adj	***		
			DP > S	***				
			SP > L	**				
			SP > P adj	***				
			SP > S	***				

Using Pearson's product-moment correlation to compare the % water within a quadrat (a visual estimation of the % of water in features), there was a clear correlation with the importance of water holding capacity on the limestone with the ecological variables of richness and abundance. This indicates the effect of beta diversity alongside the desiccation amelioration effect of more standing water. A moderately strong positive association between the abundance of sessile fauna and water holding % on limestone was found (r(28) =0.490, p=0.006) but this was not observed on basalt (r(28)= 0.357, p=0.053). Andesite was highly bimodal, either having a pool or not, with few other geomorphic features holding water. Positive correlations between water holding % and species richness were significant for basalt, limestone and andesite and positively associated but not significant on sandstone (Figure 3-11). On andesite, pools (typically higher percentage cover of quadrats) did not increase mobile abundance but were strongly correlated with increasing algae and lichen abundance, which was also found on limestone (Figure 3-11).

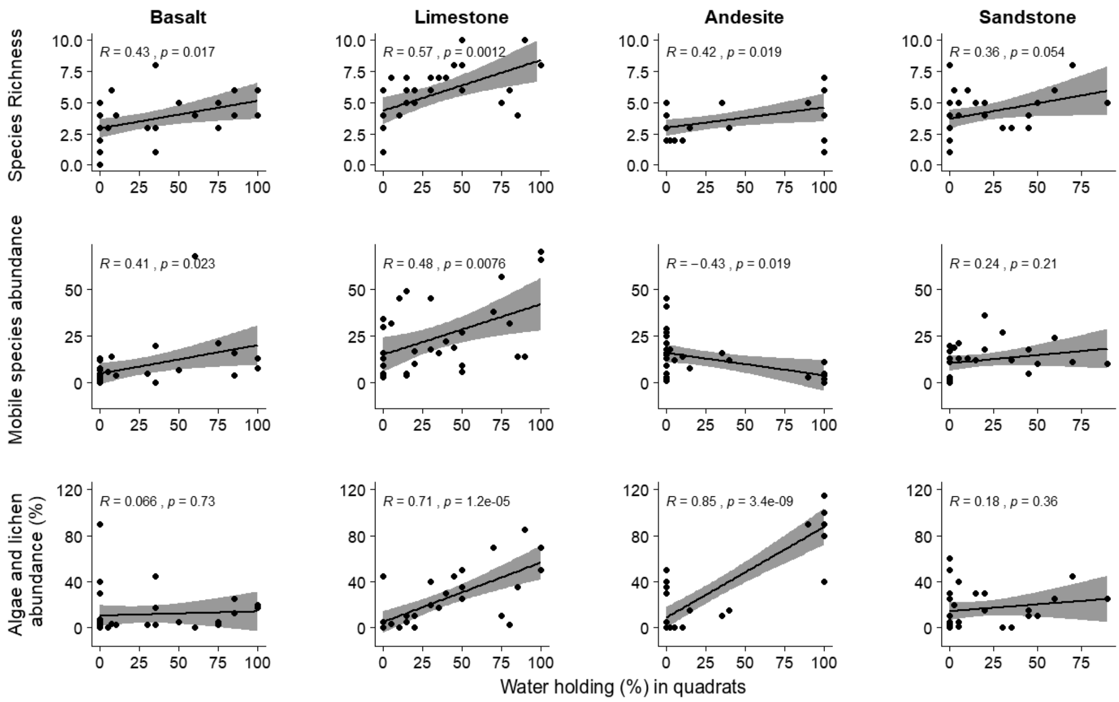

Figure 3-11. Pearson's product-moment correlation between % water holding in quadrats (25 cm^2) and species richness, mobile species abundance and algae and lichen abundance (%) on individual lithologies. A linear regression line is fitted with the grey shaded area representative of a 95% confidence interval. Each point in the scatter plot is representative of an individual quadrat.

3.7.1 Summary - Regional

In summarising the results in Table 3-10, Hypothesis 1 can be rejected as in comparing between two lithologies only (on each coast), although the softer limestone outperformed basalt in the ecological variables examined (species richness, mobile and sessile abundance and species diversity), the same cannot be said when comparing sandstone with andesite on the east coast, which only statistically exceeded the andesite in species diversity. For hypothesis 2, boring species were only found on limestone (*Polydora ciliata*) while the most well-known rock coast bioeroder (limpets) did not differ between lithologies on each coastline, as such the overall hypothesis is rejected. Both hypothesis 3 and 4 can be accepted as several microhabitats outperformed the platform surface in attracting greater species richness and abundance, with pools performing the best on average across the ecological variables examined. Finally, hypothesis 5 is accepted only on the limestone and is rejected for the other lithologies, which indicates that it is especially important to consider rock material properties in order to understand the ecological community.

Table 3-10. Support for hypotheses by ecological metrics (SR= species richness, MA = mobile abundance, SFA= sessile fauna abundance, ALA= algae and lichen abundance, D= diversity). "+" indicates a positive result in the hypothesised direction where L=limestone, B=basalt, A=andesite and S=sandstone, "-" indicates a negative response, N/A was not tested for this hypothesis and X= findings reject hypothesised direction.

	Ecological metric				
Hypothesis	SR	MA	SFA	ALA	D
[H1] Softer lithologies will have greater species richness, abundance (mobile and sessile species) and diversity than harder lithologies (softer limestone and sandstone > harder basalt and andesite)	+ (L>B)	+ (L>B)	+ (L>B)	+ (L>B)	+ (L>B) + (S>A)
[H2] Bioerosive (grazing and boring species) abundance on calcium rich lithology (limestone > basalt, andesite, sandstone)	N/A	X	+ (L>B)	N/A	N/A

[H3] Abundance and richness (microhabitats > platform surface)	+	+	+	+	N/A
[H4] Abundance and richness (pools > other microhabitats)	+	+	+	+	N/A
[H5] Higher water holding % increases richness and abundance	+ (L,B,A)	+ (L,B) - (A)	+ (L)	+ (L, A)	N/A

3.8 Results - Local comparison at Barns Ness

3.8.1 Rock material properties

Both platforms are relatively flat, with the limestone platform having more of a seaward slope than seen on the sandstone. It is evident that the sandstone is a more jointed surface, with the presence of crevices picked up by the scans as low points on the profile graph, whereas limestone has less variation between different steps/features (Figure 3-12). The mesh created in Figure 3-13 shows the presence of these crevices more clearly whereas the limestone is lacking the presence of more distinct geomorphic features. The hummocky surface (small mounds) of the limestone creates topographic relief but the overall platform surface is smooth, reducing its ecological suitability compared to the sandstone platform.

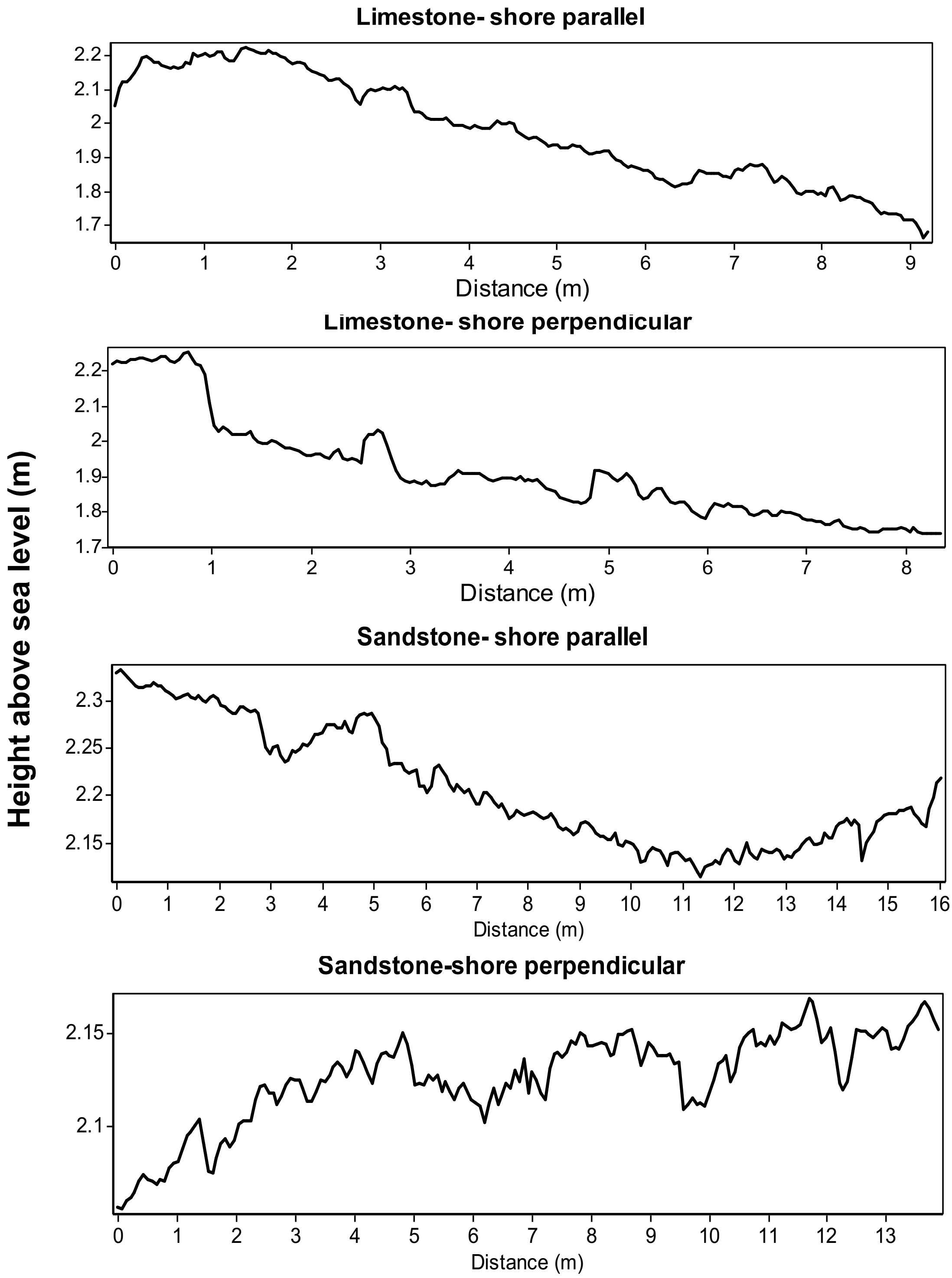

Figure 3-12. Limestone and sandstone platform surface elevation profile graphs (parallel and perpendicular to the shore) with distance along the platform and height above sea level (m).

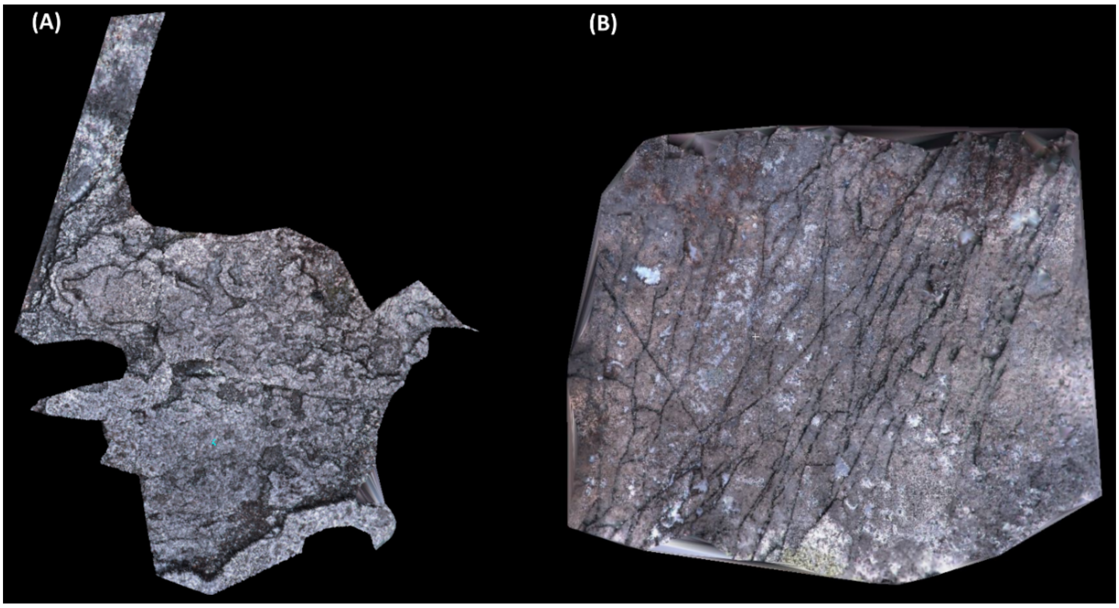

Figure 3-13. Mesh created of platform surface for (A) limestone and (B) sandstone at Barns Ness.

In comparing the two lithologies from the results of the laboratory tests, the sandstone is more ecologically suitable as it has a higher albedo ($H(1)= 97.23$, $p<0.001$), water absorption capacity ($H(1)=9.8$, $p<0.01$) and porosity ($H(1)=9.8$, $p<0.01$) than the adjacent limestone (Figure 3-14). Albedo varied within lithology as rough samples were darker than smooth for limestone ($z=6.836$, $p<0.05$) and for sandstone both rough and smooth samples were more reflective than weathered samples ($z=5.29$, $p<0.001$ and $z=3.02$, $p<0.01$ respectively) but the differences in their albedo was minor (Figure 3-14).

There was no difference in surface roughness between lithologies at the 0.25 mm^2 and 25 mm^2 scales. When comparing both smooth samples and weathered samples between lithologies, limestone was slightly rougher at the 0.25 mm^2 scale ($H(1)=3.857$, $p<0.05$ for both) and additionally for weathered samples at the 25 mm^2 scale ($H(1)=3.857$, $p<0.05$, Figure 3-14). Rough samples did not differ between lithologies at either of the examined scales.

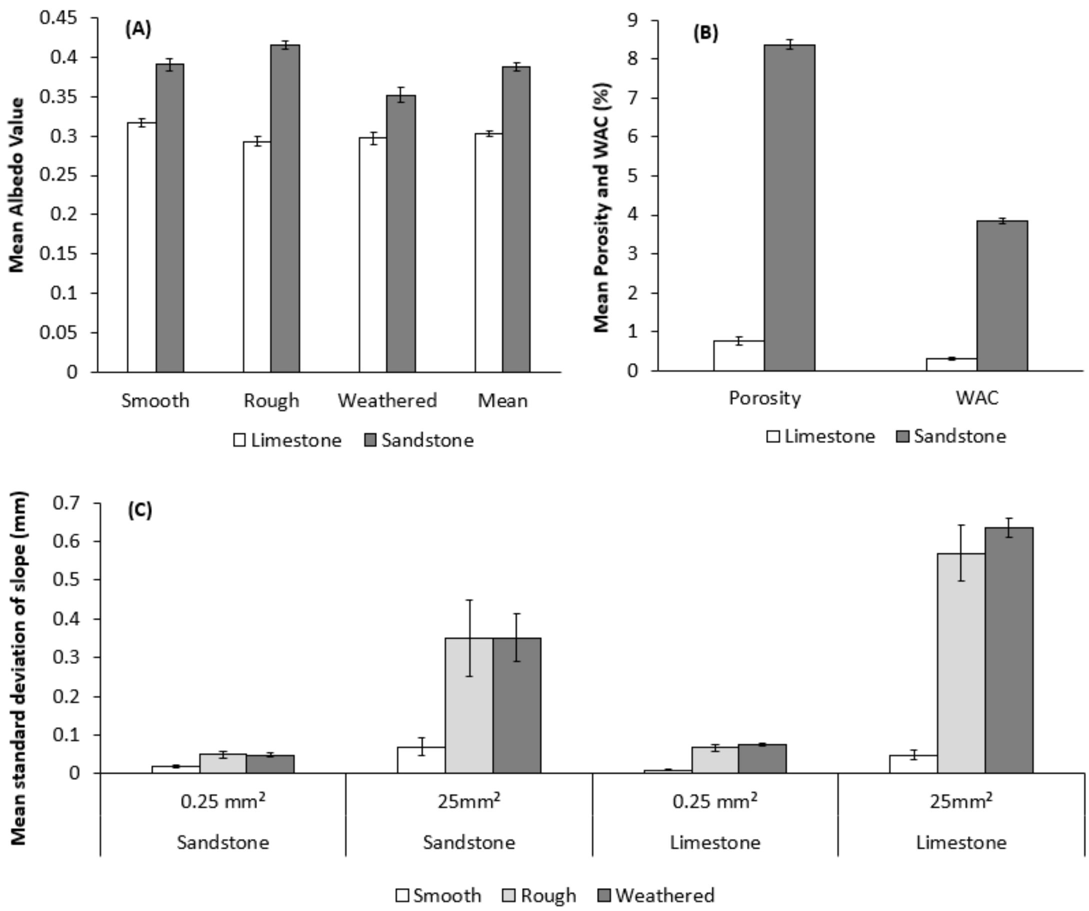

Figure 3-14. Laboratory test results for (A) mean albedo (B) mean porosity and water absorption capacity (WAC %) and (C) surface roughness at the 0.25 mm^2 and 25 mm^2 scales for sandstone and limestone for smooth, rough and weathered samples (albedo and roughness) ($\bar{x}$ ± standard error).

3.8.2 Effects of rock mass properties on species abundance

A total of ten species were recorded across the limestone and nine across the sandstone shore platforms. Two species were unique to sandstone (low abundance of *Rhodothamniella floridula* and one lone *Littorina obtusata*) and two unique to limestone (*Ulva sp.* and two common gobies in a crevice). *Semibalanus balanoides* was infrequently recorded on both limestone (1.02% ± 0.84 SE) and sandstone (0.43% ± 0.33 SE) platforms. Species richness did not significantly differ between the two lithologies, although mobile species abundance was significantly higher on sandstone (Figure 3-15, Table 3-11). Algae and lichen abundance ($H(1)=2.287$, p=0.130) and the total number of microhabitats did not significantly differ between lithologies. Although the total number of microhabitats did not differ, it is clear that the two lithologies result in different habitat types being more prevalent, with crevices more abundant (Figure 3-16) on sandstone and shallow pools and ledges more abundant on the limestone. Although the limestone has a broader range of feature types, the flatter sandstone which has a greater abundance of crevices is better for ecology (Figure 3-16). From personal observation, the limestone platform was influenced by abrasion which would limit species presence and distribution, making it less ecologically suitable.

Table 3-11. GLMs for a) species richness and b) mobile species abundance and c) total features with lithology (limestone and sedimentary) (***= $p<0.001$, NS= Not significant).

Species richness (Quasi-Poisson)	**Estimate**	**Std. Error**	**T value**	**P-value**
(Intercept)	0.624	0.138	4.600	***
Sandstone- Limestone	0.223	0.182	1.226	NS
Mobile abundance (Negative Binomial)	**Estimate**	**Std. Error**	**Z value**	**P-value**
(Intercept)	0.262	0.289	0.908	NS
Sandstone- Limestone	1.454	0.384	3.790	***
Total features (Quasi-Poisson)	**Estimate**	**Std. Error**	**T value**	**P-value**
(Intercept)	0.033	0.149	0.221	NS
Sandstone- Limestone	-0.298	0.228	-1.312	NS

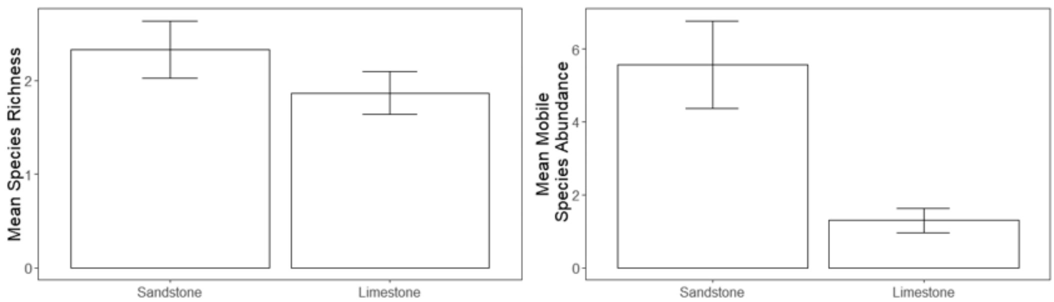

Figure 3-15. Mean species richness and mobile species abundance between lithologies at Barns Ness (per 25 cm^2, n=30 quadrats per platform) ($\bar{x}$ ± standard error).

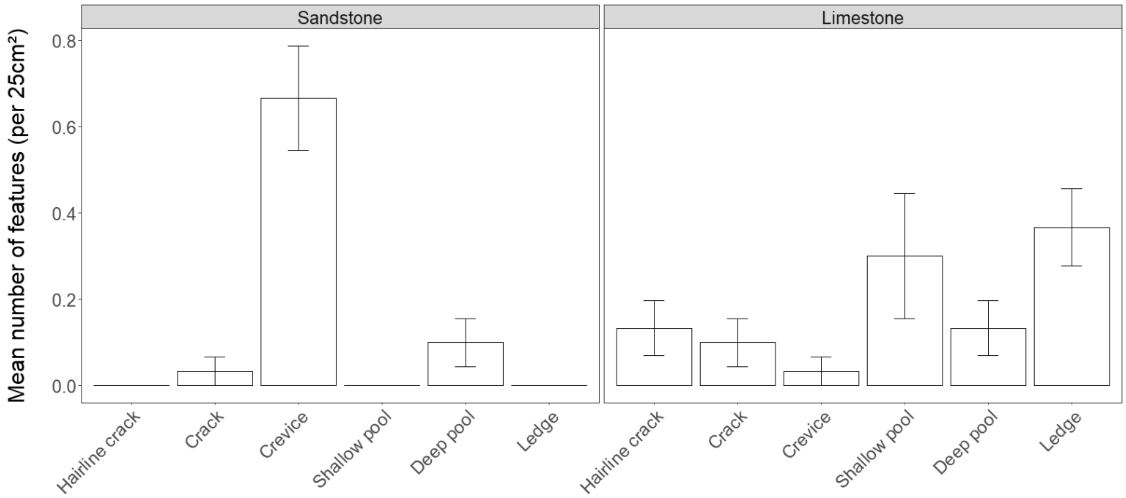

Figure 3-16. Mean number of features per quadrat (per 25 cm^2, n=30 quadrats per platform) ($\bar{x}$ ± standard error).

A new dataset was created to examine microhabitats to determine whether they exerted an influence on ecological variables with quadrats classified into those with and without features. Each individual feature within each quadrat was included and where features did not cover 100% of the quadrat area, the platform surface was included. Feature types were grouped into pool (shallow and deep), joint (crack, crevice and hairline crack) and surface to allow comparison between sites. Ledges were excluded from this initial analysis as these were only present on limestone. Limestone and joints were the reference lithology and feature type. The sandstone platform is more species rich with greater mobile abundance than the limestone platform (Table 3-12, p<0.01). This is likely a result of abrasion scouring the platform surface on the limestone (personal observation).

Table 3-12. GLMs for a) species richness and b) mobile species abundance in relation to lithology (limestone and sedimentary) and geomorphic feature (joint, pool, surface) (***= p<0.001, **=p<0.01, *=p<0.05, NS= Not significant).

Species richness (Quasi-Poisson)	**Estimate**	**Std. Error**	**T value**	**P-value**
(Intercept)	-0.693	0.566	-1.224	NS
Sandstone	1.773	0.587	3.018	**
Feature-Pool	0.588	0.681	0.863	NS
Feature-Surface	0.963	0.596	1.618	NS
Sandstone: Feature- Pool	-0.820	0.819	-1.002	NS
Sandstone: Feature- Surface	-2.611	0.673	-3.874	***

Mobile abundance (Negative Binomial)	Estimate	Std. Error	Z value	P-value
(Intercept)	-0.981	0.722	-1.359	NS
Sandstone	2.903	0.783	3.708	***
Feature-Pool	0.624	0.902	0.692	NS
Feature-Surface	0.447	0.795	0.562	NS
Sandstone: Feature- Pool	-0.118	1.198	-0.099	NS
Sandstone: Feature- Surface	-3.467	0.935	-3.710	***

In examining species richness on sandstone, the platform surface had lower species richness ($z=-5.284$, $p<0.001$) and mobile species abundance ($z=-7.709$, $p<0.001$) than crevice habitat (Figure 3-17). The platform surface also had lower species richness ($z=-2.759$, $p<0.05$) and mobile abundance ($z=-6.109$, $p<0.001$) than deep pools. Although algae and lichen abundance (%) differed between features ($H(3)=12.657$, $p<0.01$), post-hoc comparisons did not show a difference between geomorphic feature types.

On limestone, species richness, mobile species abundance and algae and lichen abundance did not significantly differ between geomorphic habitat types, with habitat features exerting limited influence on the ecology. Abrasion by rocks and stones was prevalent on sections of the limestone platform and the high abundance of shallow features (shallow pools, hairline cracks, cracks) would be susceptible to repeat disturbance from abrasion, limiting the ecological significance of these geomorphic features compared to the sandstone where abrasion was not observed. Algae and lichen abundance was however, notably greater on the platform surface than in other habitat types (Figure 3-17).

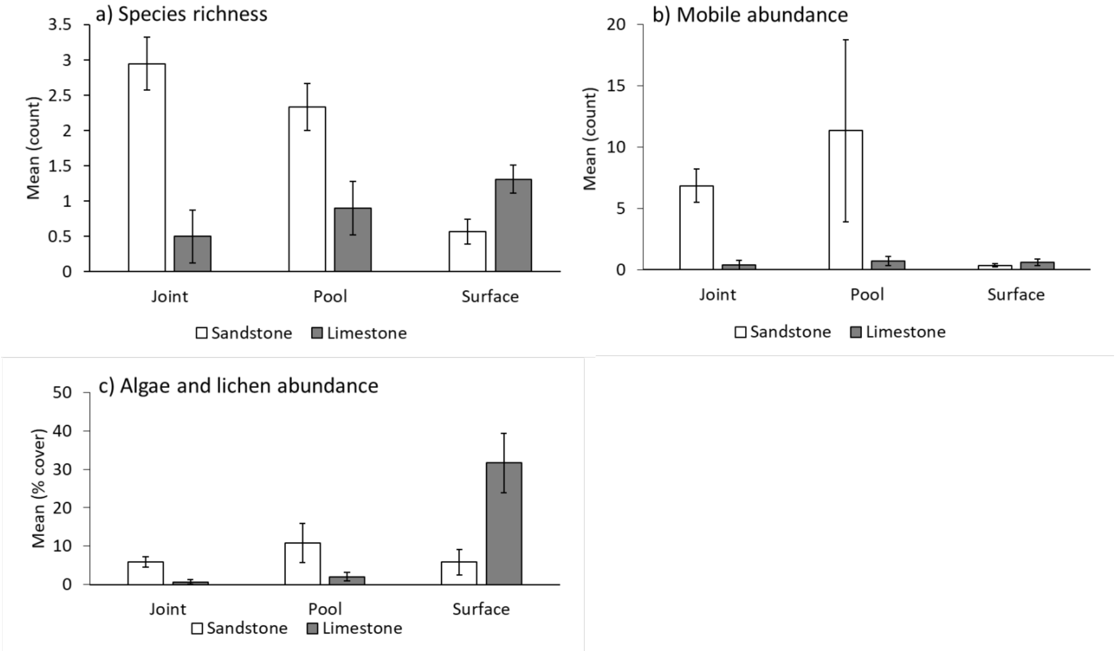

Figure 3-17. Mean a) species richness, b) mobile species abundance and c) algae and lichen abundance in features on the sandstone and limestone platforms at Barns Ness. Sandstone joint n=18, pool n=3, surface n=30, limestone joint n=8, pool n=10, surface n=29. ($\bar{x}$ ± standard error).

3.9 Results- Barns Ness habitat mapping on sandstone platform

Creating a 'hotspot' map of the shore platform using the DGPS points and quadrat data highlights the apparent association between species richness and water holding (%), with several high density (4-7 species) areas matching the locations where water holding (%) exceeds 30.59% (Figure 3-18). Mobile abundance does not show much of an association with the presence of water but visually appears to have a moderate association with the total number of features available, which were predominately crevices on this platform (Figure 3-13,Figure 3-18).

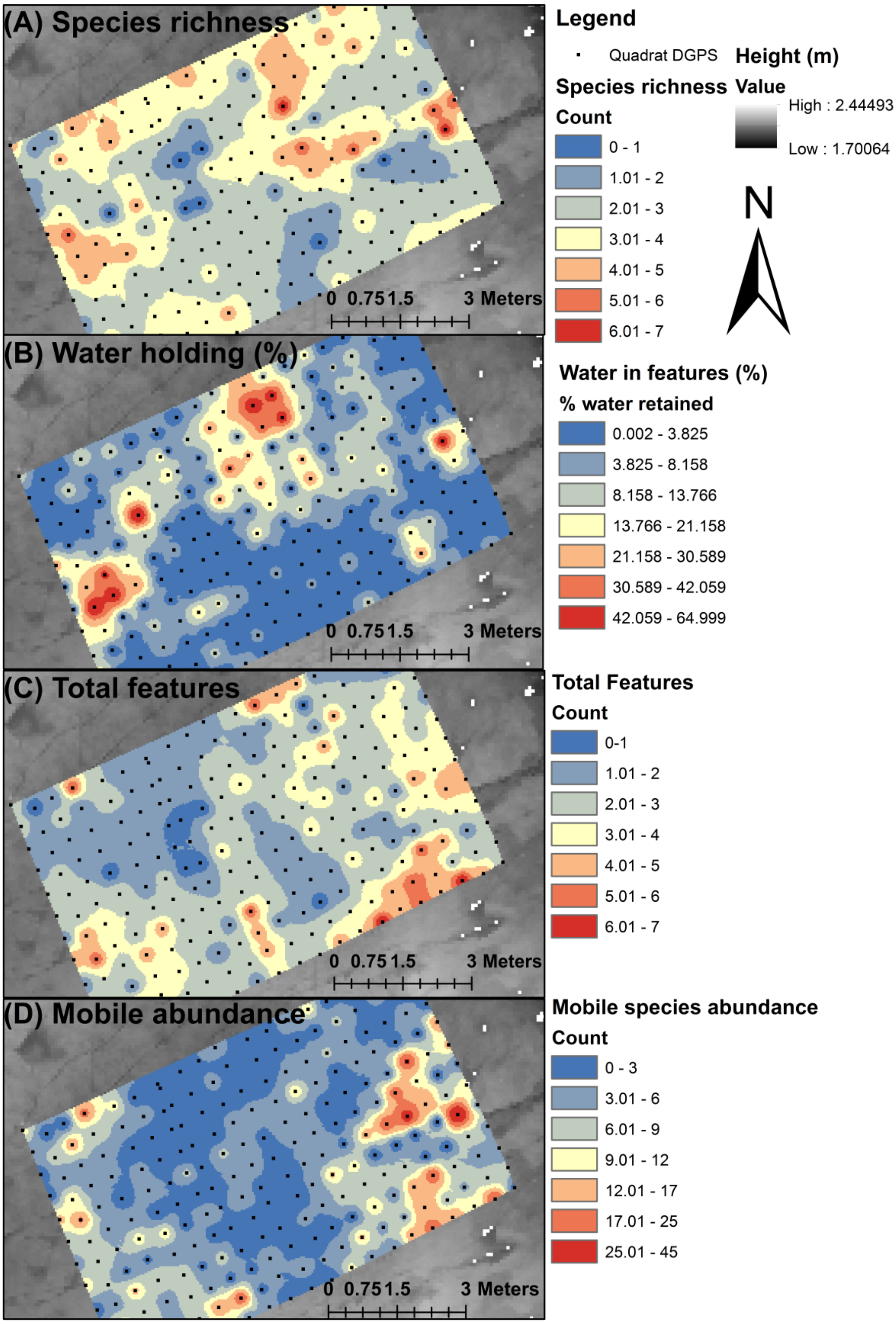

Figure 3-18. ‘Hotspots’ created using n=240 quadrats (50 cm^2) over a 10x6 metre area of the sandstone platform at Barns Ness for (A) species richness, (B) water holding (%), (C) total features and (D) mobile species abundance.

Pearson's product-moment correlation was used to identify whether there was a linear relationship between the above patterns. Species richness and algae and lichen abundance (%) were positively associated with water held in features (%) (visual estimation of the % of water in features). Species richness also exhibited a positive association with the total number of geomorphic features available but this was not true for algae and lichen species abundance (Figure 3-19).

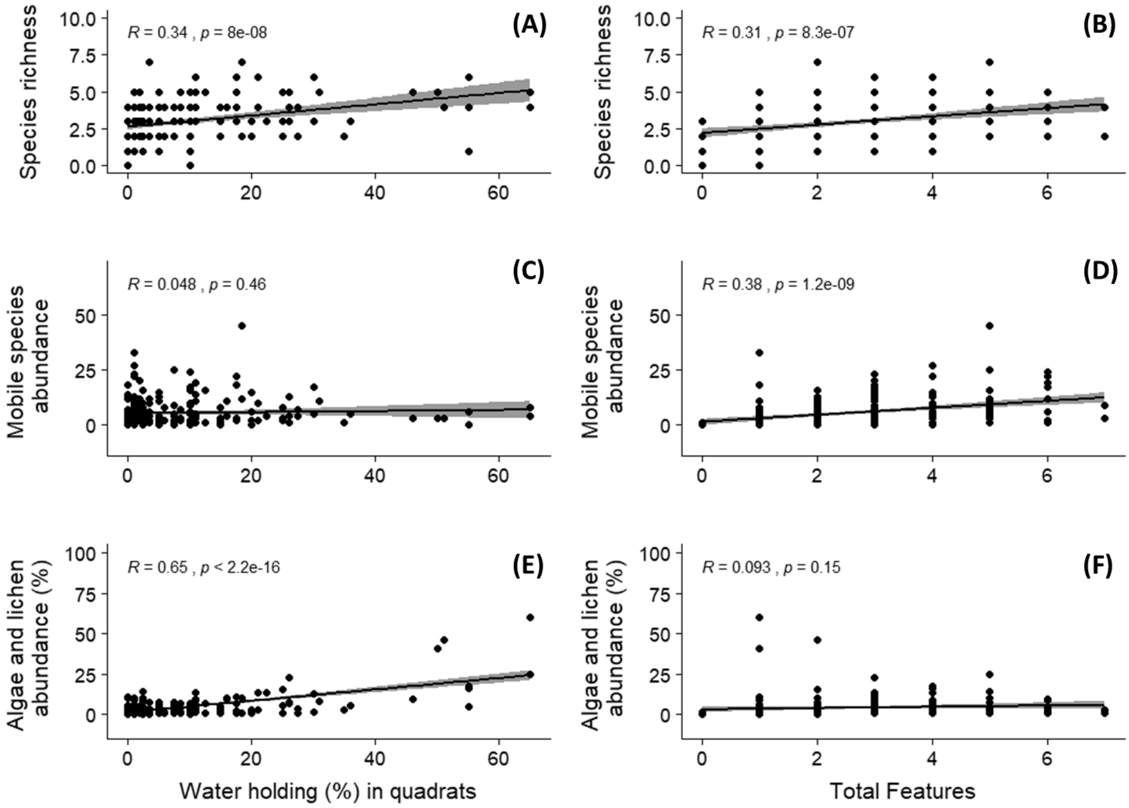

Figure 3-19. Pearson's product-moment correlation between (A-B) species richness, (C-D) mobile species abundance, (E-F) algae and lichen abundance (%) and (A,C,E) water holding (%) in quadrats (50 cm^2) and (B,D,F) total geomorphic features (count) on the sandstone platform at Barns Ness (n=240 quadrats). The grey shaded area is representative of a 95% confidence interval and each point represents an individual quadrat.

Individual geomorphic features were sub-divided into pools (shallow and deep, n=146), joints (cracks and crevices, n=268) and ledges (n=46). Pits and hairline cracks were also examined but were shown not to have a relationship with species richness or abundance. Only n=3 pits were recorded and so were too few for individual analysis. A total of 149 hairline cracks were recorded, which did not influence species richness (R=-

0.052, p=0.41), mobile species abundance (R=0.12, p=0.065) or algae and lichen abundance (R=-0.05, p=0.44).

The abundance of mobile species and overall species richness moderately increased with the presence of joints, with a relatively strong association between jointing and mobile abundance (Figure 3-20). Pools had no significant influence on mobile species abundance but both species richness and algae and lichen abundance were found to have a moderate positive association with the abundance of pools. Ledges were only found to moderately influence mobile species abundance (Figure 3-20).

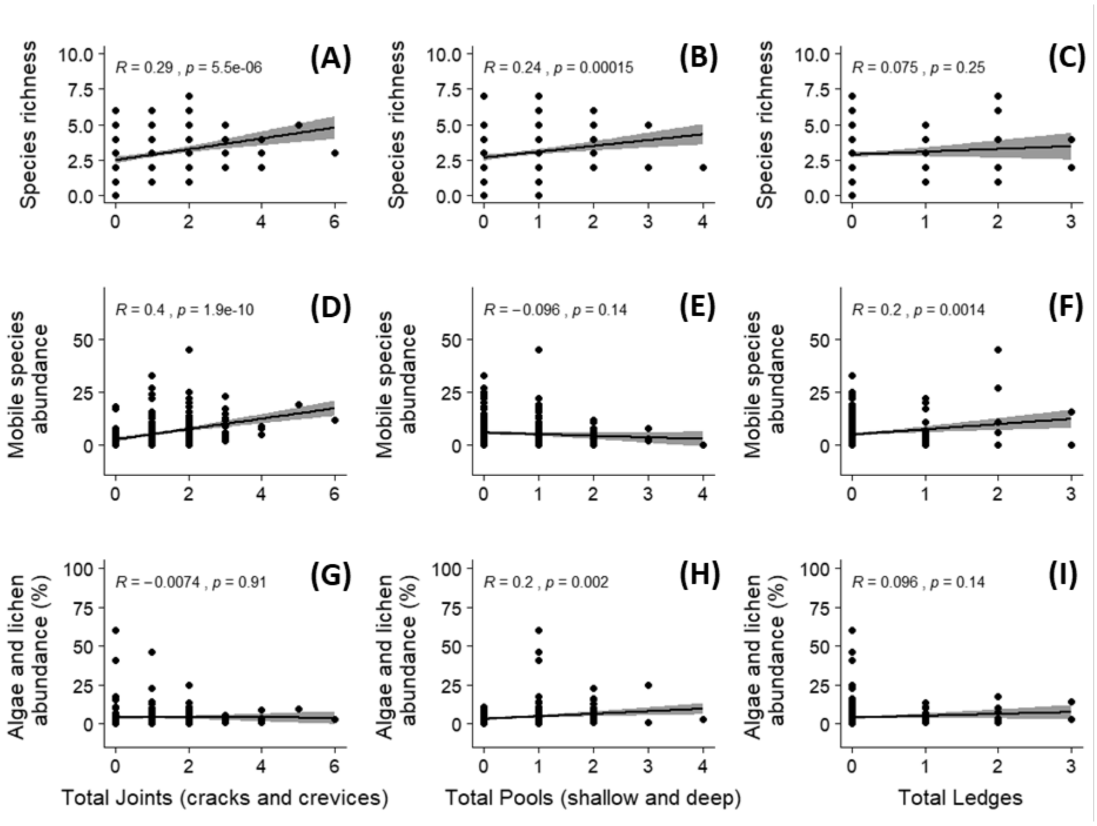

Figure 3-20. Pearson's product-moment correlation between (A-C) total species richness, (D-F) total mobile species abundance, (G-I) total algae and lichen abundance (%) and (A,D,G) total joints, (B,E,H) total pools and (C,F,I) total ledges (count) on the sandstone platform at Barns Ness (n=240 quadrats). The grey shaded area is representative of a 95% confidence interval and each point represents an individual quadrat.

3.9.1 Summary- Barns Ness

More porous lithologies have been found to remain wetter for longer, creating a damper microclimate that reduces desiccation risk (Coombes and Naylor, 2012). Lighter colours reduce thermal stress for species, as darker substrates have higher temperatures at low tide and can negatively impact the survival and abundance of species (Kordas et al., 2014). The Barns Ness sandstone was lighter in colour and more porous than the limestone, making these parameters at this site more ecologically suitable on the sandstone and allowing the confirmation of hypothesis 6. Surface roughness did not differ between the two lithologies at the mm-scale except for smooth and weathered samples of limestone being rougher than their equivalents in sandstone. The properties of the sandstone may have contributed to its ecological suitability in the field (greater species richness on average and significantly higher mobile species abundance). This fulfils the criteria for hypothesis 7, that the most ecologically suitable lithology from lab tests will have greater species richness and mobile abundance in the field, allowing the acceptance of hypothesis 7.

In focusing on geomorphic feature types, both species richness and mobile species abundance was greater in crevice habitat on the sandstone than the platform surface, with deep pools also exceeding the surface in attracting high numbers of mobile species. This is not true of the limestone platform as shallow pools and ledge habitats did not match the platform surface in terms of species richness, likely due to the high abundance of algae on the surface. Features on the studied limestone also made little difference in altering mobile species abundance. This results in hypothesis 8 and 9, that higher numbers and a greater variety of species will be found in microhabitats than the surface and that pools and crevices will be the most successful of these microhabitats, being accepted for the sandstone platform and rejected for the limestone platform.

Finally, in examining the model of the sandstone platform, hypothesis 10, that a higher percentage of water held (%) significantly increases species abundance and richness, is rejected. Although there is a positive association between species richness and water availability, this is not true of mobile species abundance where the density of features is more important. Joints, pools and ledges were noted as important habitats on this shore for species richness and abundance.

3.10 Results- Glamorgan

A greater number of geomorphic feature types were found in the more densely jointed area (Table 3-13), confirming the valid selection of the survey areas. As sites were selected based on the density of jointing, it is not surprising that only crevices were found in higher occurrence in the more densely jointed area (t=2.384, p<0.05, Figure 3-21), confirming Hypothesis 11 that joint density can vary within the same bed layer.

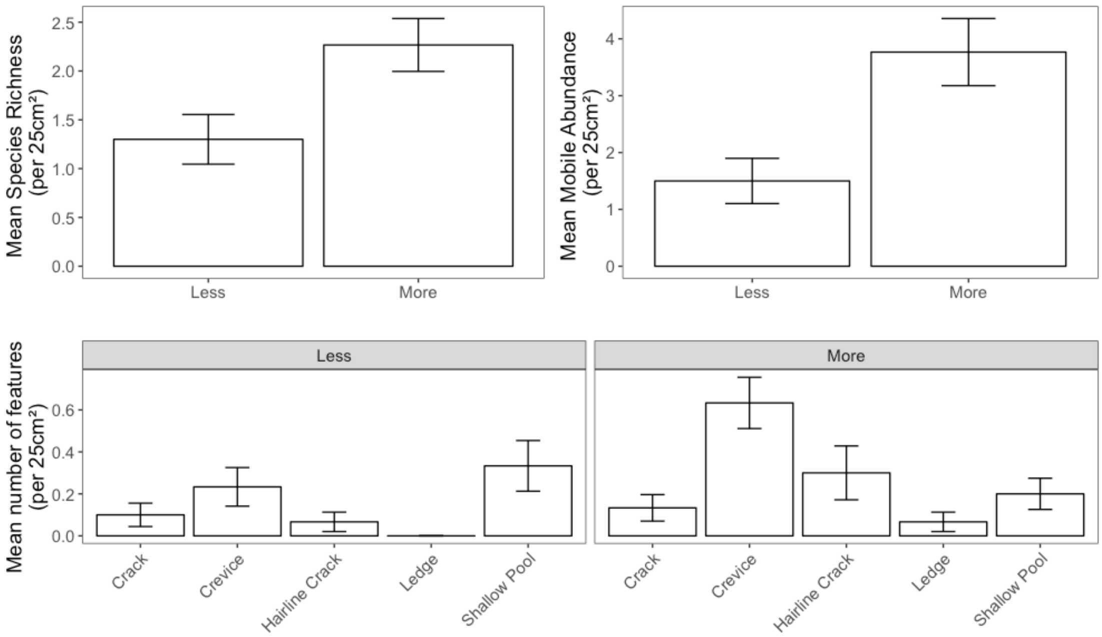

Figure 3-21. Mean species richness, mobile species abundance (count) and number of features (habitat types) per 25 cm^2 in less and more densely jointed survey areas ($\bar{x}$ ± standard error).

A total of nine species were recorded across the surveys in the more densely jointed area compared to six species in the less densely jointed area. Most species observed were present across both survey areas, with *Chondrus crispus* (0.5%, n=1 quadrat), *Polydora ciliata* (2.3 ± 1.69 SE) and *Nucella lapillus* (1, n=1 quadrat) uniquely present in low abundances in the more densely jointed area. *Chthamalus montagui* was present in both survey areas in low abundance (0.25% ± 0.05 SE, low density, 0.87% ± 0.21 SE, high density). Joint density can be seen to positively influence species richness (p<0.01) and mobile abundance (p<0.01) (Table 3-13, Figure 3-21), particularly for *Patella vulgata* (p<0.001) (Figure 3-22. This allows Hypothesis 12, that a more densely jointed area will have greater numbers of crevices and these microhabitats will increase richness and abundance of species compared with a less densely jointed area, to be confirmed.

Table 3-13. Summary of results for Negative Binomial (NB) and Quasi-Poisson (QP) GLM for total species richness, mobile abundance, *Patella vulgata* and Total Features.

	Total SR (NB)				Mobile abundance (NB)			
	Estimate	**Std. Error**	**Z value**	**P**	**Estimate**	**Std. Error**	**Z value**	**P**
Intercept	0.262	0.168	1.566	NS	0.401	0.245	1.655	NS
More jointed-Less jointed	0.556	0.213	2.615	**	0.921	0.327	2.818	**
	***Patella vulgata* (NB)**				**Total Features (QP)**			
	Estimate	**Std. Error**	**Z value**	**P**	**Estimate**	**Std. Error**	**T value**	**P**
Intercept	-0.762	0.360	-2.118	*	-0.310	0.181	-1.717	NS
More jointed-Less jointed	1.692	0.448	3.776	***	0.598	0.225	2.658	*

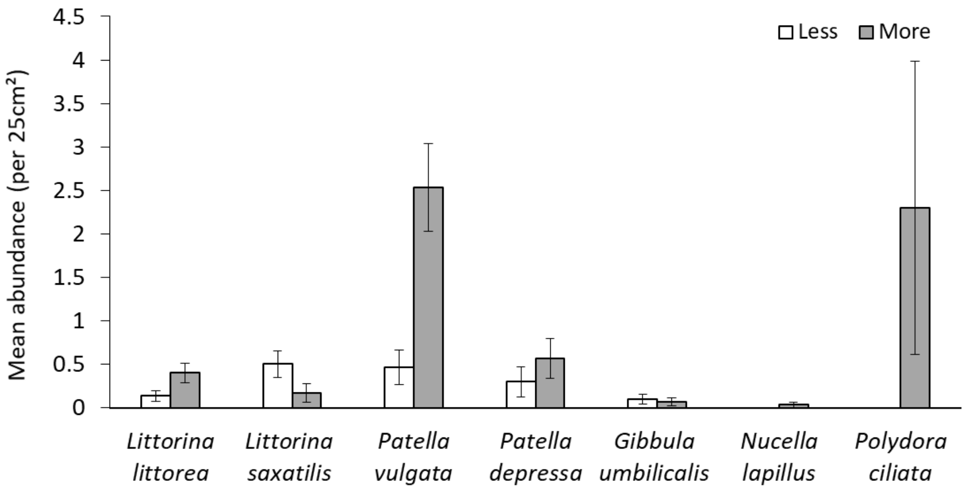

Figure 3-22. Mean abundance of mobile species in a less and more densely jointed area within the same bed layer (x̅ ± standard error).

Although species richness was higher in the more densely jointed area, no one habitat can be attributed to significantly influencing this higher species richness as significant results were only found between several habitat types and the platform surface (Table 3-14). This is also true of the less densely jointed area, with known refugia outperforming the platform surface but no other habitat types (Table 3-14).

Table 3-14. Significant GLM post-hoc comparisons for species richness between geomorphic features in the (A) less densely jointed and (B) more densely jointed survey area.

(A) Less densely jointed – Quasi-Poisson					
	Estimate	**Std. Error**	**T value**	**Finding**	**P-value**
Surface – Shallow pool	-1.322	0.383	-3.453	Shallow pool > Surface	**
Crevice – Surface	1.977	0.352	5.615	Crevice > Surface	***
(B) More densely jointed – Quasi-Poisson					
	Estimate	**Std. Error**	**T value**	**Finding**	**P-value**
Shallow pool – Surface	1.124	0.379	2.969	Shallow pool > Surface	*
Crevice – Surface	1.322	0.286	4.620	Crevice > Surface	***
Pool adjacent – Surface	1.347	0.354	3.802	Pool adjacent > Surface	**
Ledge – Surface	1.753	0.443	3.954	Ledge > Surface	**

There was no difference in the abundance of mobile species present on the platform surface between both survey areas as only a few *Patella vulgata* were observed outside of features (n=1, less jointed and n=7 from 3 quadrats, more jointed). The only other species present on the platform surface was a low cover of *Chthamalus montagui.* In the more densely jointed area, crevices were not the frontrunner for mobile species abundance as they did not differ in abundance from other cm-scale habitat features but they did attract higher numbers of mobile species than the platform surface and hairline cracks (Table 3-15), which add very minimal habitat value. In the less densely jointed area, both crevice and shallow pool habitat was more attractive to mobile species than the platform surface (Table 3-15, p<0.01). In addition to this, crevices attracted a greater abundance of mobile species than shallow pools and pool adjacent areas (halo of wetted surface around the pool) (Table 3-15, p<0.05). As crevices and shallow pools had higher species richness and mobile species abundance than adjacent areas on the platform surfaces, Hypothesis 13 can be accepted.

Table 3-15. Significant GLM post-hoc comparisons for mobile species abundance between geomorphic features in the (A) less densely jointed and (B) more densely jointed survey area.

(A) Less densely jointed- Quasi-Poisson					
	Estimate	**Std. Error**	**Z value**	**Finding**	**P-value**
Surface – Shallow pool	-3.887	1.090	-3.566	SP > Surface	**
Crevice – Shallow pool	0.981	0.357	2.749	Crevice > Shallow pool	*
Crevice – Surface	4.868	1.070	3.213	Crevice > Surface	***
Pool adjacent – Crevice	-1.936	5.129	-3.775	Crevice > Pool adjacent	**
(B) More densely jointed- Negative Binomial					
	Estimate	**Std. Error**	**Z value**	**Finding**	**P-value**
Hairline Crack – Crevice	-2.110	0.707	-2.987	Crevice > Hairline crack	*
Crevice – Surface	2.872	0.470	6.109	Crevice > Surface	***
Ledge – Surface	2.554	0.813	3.143	Ledge > Surface	*
Pool adjacent – Surface	2.659	0.570	4.669	Pool adjacent > Surface	***
Shallow pool – Surface	1.966	0.612	3.213	Shallow pool > Surface	*

Table 3-16. Mean crevice feature dimensions (mm) and water-holding (%) (± SE).

	Water-holding (%)	**Width (mm)**	**Depth (mm)**
Less jointed	5.83 ± 2.93 SE	55.00 ± 9.13 SE	45.83 ± 6.11 SE
More jointed	3.19 ± 1.27 SE	45.47 ± 4.32 SE	38.28 ± 3.15 SE
Average	3.91 ± 1.21 SE	48.07 ± 4.00 SE	40.34 ± 2.85 SE

The crevice dimensions at this site appear optimally suited to provide sufficient habitat for larger species such as limpets, offering a slightly more wetted and sheltered habitat (from both desiccation and potential abrasion) than found on the platform surface (Table 3-16).

Except for *Polydora ciliata* (n=50) in a single water holding crevice in the more densely jointed area, all other species recorded in crevices were from the Gastropoda class. A lone *Nucella lapillus* was recorded in a single crevice in the more densely jointed area and a single *Gibbula umbilicalis* was recorded once in a crevice in the more densely jointed area and another in the less densely jointed area. In examining the 16 crevices in the more densely jointed area and the 6 in the less densely jointed area, *Patella vulgata* was the most commonly occurring species in the crevices (more= 2.75 ± 0.70 SE, less=± 0.68 SE per crevice), with *Patella depressa* (more= 0.69 ± 0.35, less= 0.67 ± 0.67

SE), *Littorina littorea* (more=0.44 ± 0.16 SE, less= 0.5 ± 0.22 SE) and *Littorina saxatilis* (more= 0.13 ± 0.09 SE, less = 1 ± 0.45 SE) also occurring frequently.

3.11 Body size and features

Negative Binomial GLMs were conducted on gastropod abundance in features, excluding for the influence of lithology (grouped all site data). Gastropods were the focal species here as these species (particularly *Patella vulgata* and *Littorina littorea*) were relatively ubiquitous across all shores and habitat types, frequenting pools and crevices (Table A 3-3). The body size of juvenile and adult gastropods is well suited to the dimensions of crevices and pool habitats, which allow species room to grow and provide refuge. These species range in size from mm-cms and so outline the suitability of habitats for a variety of body sizes. Surface habitat does not offer suitable habitat provision for Gastropods which frequent more sheltered refugia (Table A 3-4). Deep pools and crevice habitat were consistently outperforming smaller scale features in their provision of effective habitat for these species (Table A 3-4). Cracks (<2.5 cm in width) were ineffective compared to larger scale habitat, this is likely as these features do not have an appropriate width to host the body sizes of mobile intertidal species.

3.12 Results- Summary

Overall findings highlight the importance that microhabitats have in influencing the distribution of species and increasing species richness. There are also notable differences between coast, lithologies and the microhabitats within them (Table 3-17).

Table 3-17. Summary of chapter hypotheses.

Hypotheses		Outcome from this research	Summary decision
[R1-H1]	Softer lithologies (limestone and sandstone) will have greater species richness, abundance (mobile and sessile species) and diversity than harder lithologies (basalt and andesite)	Softer limestone was better than basalt, but sandstone did not perform significantly better than harder lithologies.	X
[R1-H2]	A higher abundance of bioerosive (grazing and boring) species will be found on calcium rich rock (limestone) than lithologies without calcium (basalt and andesite)	True for boring species, which were only found on the limestone at Girvan but limpet abundance did not differ between lithologies.	X

[R2-H3]	Higher numbers and a greater variety of species will be found in microhabitats than on adjacent surfaces	Several microhabitats outperformed the platform surface on multiple lithologies	✓
[R2/R3-H4]	Pools will have greater species richness and abundance than other microhabitats.	Deep pools performed the best on average where they were present.	✓
[R2/R3-H5]	A higher percentage of water held significantly increases species abundance and richness.	This is significant on the examined limestone at Girvan but not for other lithologies.	✓ (limestone) ✗ (other examined lithologies)
[R1-H6]	The Barns Ness sandstone will be more ecologically suitable (lighter colour, more porous) than the limestone.	Although limestone has greater fine-scale roughness, for other parameters i.e. colour and porosity, sandstone is more ecologically suitable	✓
[R1- H7]	The most ecologically suitable rock from laboratory testing will have greater species richness and mobile species abundance in the field.	Sandstone had higher species richness and mobile abundance on average than the adjacent limestone platform. This may be due to having a higher number of refugia but the underlying lithology is also more ecologically suitable.	✓
[R2-H8]	Higher numbers and a greater variety of species will be found in microhabitats than on adjacent surfaces	Species richness and abundance was greater in features than the platform surfaces on the sandstone. Features made little influence on the limestone platform.	✓ (sandstone) ✗ (limestone)
[R2/R3-H9]	Pools and crevices (more abundant on the sandstone from ground truthing) will have greater species richness and abundance than other microhabitats	Deep pools and crevices were the most effective microhabitat on the sandstone, but these features made little influence on the limestone platform.	✓ (sandstone) ✗ (limestone)
[R2/3-H10]	A higher percentage of water held significantly increases species abundance and richness.	There is a positive association between species richness and water availability but not with mobile species, crevices appear more important for these species.	✗

[H11]	Density of jointing can differ within the same bed layer.	There is a significant difference in joint density between two adjacent areas of the shore platform.	✓
[R2-H12]	A more densely jointed area will have greater numbers of crevices and these microhabitats will increase richness and abundance of species compared with a less densely jointed area.	Joint density positively influenced species richness and mobile abundance.	✓
[R2/R3-H13]	Crevices and pools have higher species richness and mobile species abundance than adjacent areas on the platform surface.	Crevices and shallow pools met these criteria, with very few species present on the surface.	✓

3.13 Discussion

At the regional and national scale (10s-100s of km), the influence of lithology does not seem to matter (Chapter 2), but by incorporating geomorphic features and examining individual sites there is a clear influence of geomorphic complexity, rock mass and rock material properties on the distribution, abundance and richness of species. In comparing multiple lithologies, geomorphic complexity (arising from process interactions with rock material and rock mass properties) matters ecologically and within individual lithologies, such as at Glamorgan, the density of features also exerts an influence on species distributions. The abundance of pools and ledges at the Girvan site and the resulting high diversity, richness and abundance of species means that it could be proposed that more biodiversity would be expected on lithologies that generate greater habitat complexity, such as softer, calcium-based lithologies.

This requirement for softer, calcium-based lithologies is particularly true with regards to sessile fauna abundance, especially with the boring *Polydora ciliata* which was present on calcium rich lithologies in high abundance on the limestone at Girvan and in low abundance on the limestone at Glamorgan. These boring species have further potential in altering the porosity of the rock by breaking up the rock at a micro-scale, although this is typically only when present in considerably high densities (such as in pools and along joints (Naylor et al., 2012b)), they still contribute to the bioerosion of the platforms where they are present and make the rock more susceptible to wave erosion (Coombes, 2014). Surveys conducted at Glamorgan as part of this thesis show that bioerosive animals (i.e. grazing limpets) congregate in places of weakness (crevices).

Boccardia sp., another bioerosive species, have also previously been found in high abundance in joints on the shore platform at this site (Naylor et al., 2012b). Where softer lithologies (suitable for bioerosion) are present and where biology is living in crevices and they are known to be bioeroders, then it shows that the biology is one of the parameters facilitating joint weakening and shaping these features. These findings contribute to the biogeomorphological understanding of how µm-mm scale erosion from individual species influences the development of cm-m scale geomorphic features on softer shore platform lithologies. The research for this thesis solely focused on the macro-scale of ecology due to the extensive nature of surveys, but it is also important to understand the microbiology on each platform. Microbial communities are fundamental in their contribution to the development of ecological communities in that they both contribute to primary productivity and create settlement cues for other species (Coombes, 2014; Hutchinson et al., 2006).

Across all shores, geomorphic features that held water or had greater relief, primarily deep pools and crevices, were the most important habitat types for species. Species richness and abundance was significantly greater where deep pools were present compared to less complex features and the platform surface. At each site, features were fundamental in influencing the distribution of species during periods of low tide. Where microhabitats were lacking in previous studies, species, such as whelks, were found to be unable to remain without shelter for prolonged periods of time and frequented crevices, which are highly beneficial as refugia (Fairweather, 1988b). Understanding the differences in shelter provision that microhabitats offer, which were most pronounced at the cm-dm scale in this research with the presence of deep pools and crevices, and the variability in microhabitat distribution on the shore will aid in the prediction of species distribution in the mid-upper intertidal zone (Crowe, 1996). At Glamorgan in particular, which is a highly wave exposed platform, with a high tidal range (and thus a high period of exposure at low tide) and a south-facing orientation, crevices were significantly more populated than the surrounding areas and have a very clear function for providing shelter.

These findings are consistent with other studies that highlighted the importance of topographic features for intertidal species by providing refuge and reducing thermal stress, desiccation stress, predation risk and wave exposure (Gray and Hodgson, 2004;

Judge et al., 2009; Strain et al., 2017a; Underwood and Jernakoff, 1984). These stressors will influence small-scale variability in the distribution of species and their utilisation of these microhabitats, depending on the physiological tolerances of individual species, as the platform surface will host more extreme temperatures (Judge et al., 2009; Miller et al., 2009) than crevices or pools.

Small-scale variation (cm-m) in abundance has previously been reported as a result of the behavioural responses of organisms to patches of microhabitats, whereas variation in abundance at the scale of 100s of m and between shores (10s of km apart), which was found to be similar, was more likely resultant from variability in processes, such as recruitment and mortality (Underwood and Chapman, 1996). However lithological differences between shores (10s of km) alter the abundance of different microhabitats and the overall topographic complexity of the shore. Where lithology is overlooked in ecological studies, such as Underwood and Chapman (1996) focusing solely on sandstone shores in determining multi-scale patterns of abundance, an important driver of structural complexity is missed out, which from this research is shown to generate greater variation in species abundance and richness between shores.

Further experimentation would be required to extract the effects of rock material influence (specifically porosity, rock chemistry and albedo) on each shore in order to further separate this from the influence of rock mass features. It is clear that lithology and local geomorphological processes that shape microhabitats exerts a statistically significant influence on intertidal species in this study as where geomorphic feature types (i.e. ledges and deep pools) were relatively equal, such as on the basalt and limestone shores in the regional comparison, there was still a significant difference in species richness and abundance, with the limestone exceeding the basalt across all variables. This supports results from previous studies that noted substrate type influenced ecological communities (Cacabelos et al., 2016; Coombes, 2011; Herbert and Hawkins, 2006), with variations in surface roughness and rock chemistry key in promoting early colonisation and varying community composition (Cacabelos et al., 2016; Coombes et al., 2011; Herbert and Hawkins, 2006). From the surveys conducted as part of this research, it can be determined that a combination of deep pools, ledges and crevices offer the best habitat for optimising biodiversity (species richness and abundance).

Higher substratum roughness also contributes to reducing environmental stressors, such as desiccation risk during periods of low tide (Coombes, 2011; Coombes et al., 2015; Sempere-Valverde et al., 2018). Certain lithologies may generate a more complex shore, such as with the limestone at Girvan whereby even within features such as ledges there was sub-features (i.e. heterogeneity <5 mm in scale, personal observation). This micro-heterogeneity within features can promote increased settlement and survival of species by increasing surface area and microhabitat diversity, providing additional attachment space for species (Firth et al., 2014b; Sempere-Valverde et al., 2018).

The combination of TLS with ecological surveys offers a basis for mapping ecological processes that could be further integrated with developing technologies and improved ease of scanning. However, although TLS provided mm-scale resolution data of the platforms it cannot capture the range of habitat complexity on the shore or the specific depth and width of all features that allow categorisation by feature type. Further to this, TLS has technical restrictions in its ability to capture standing water (Hollenbeck et al., 2014), which was required for understanding species habitat requirements on rocky shores. The data provided is accurate at the scale required for understanding species distribution on the shore but could not be properly compared with surface roughness as the high-resolution data smoothed out the topography of the shore when converted to a DEM, which underestimates the complexity of the shore (Hollenbeck et al., 2014).

With this noted in addition to time constraints, scans were only conducted at Barns Ness as given the breadth and scope of the extensive intertidal surveys at other sites in addition to other works within this thesis, there was not enough time to both manually sample and use technology to scan every shore platform. The analogue approach methodology adopted in this thesis has the advantage in capturing the dimensions of features that DEMs from TLS scans would smooth out. However, it fails to capture the rugosity and surface complexity between features, which is important for the ecology, particularly on complex shores where the full scope of features could not be fully captured by quadrat sampling.

3.14 Conclusion

The sampling strategy employed with this research offers a much more detailed insight into the importance of habitat complexity for rocky shore species than traditional ecological and biogeomorphological sampling typically allows. The research questions for this work (Section 3.2) aimed to determine whether rock material and rock mass properties influenced species richness and abundance and which microhabitats were the most important for intertidal ecology. In surveying rocky shores, this work highlighted that it is critical to account for the presence of geomorphic features and associated rock mass properties to better understand the distribution of species and their habitat requirements. This is especially important if surveys are conducted in the mid-upper intertidal zone, where these features would exert more of an influence and where their presence is more crucial for the survival of species.

Variations in species richness, abundance and diversity between lithologies showcase that more geomorphologically complex lithologies can be assumed to have higher biodiversity, with the variation of microhabitats available attracting a greater range of species. Across all examined sites, the presence of larger scale (cm-dm’s) geomorphic features, particularly deep pools but also crevices and ledges at several sites offer the optimal habitat for attracting richness and abundance of species. Overhangs were infrequently recorded but where present hosted high numbers of mobile species due to the shelter and moister microclimate they provide. This indicates that features at the scale of cm-dm’s are required on artificial shores to optimise for biodiversity. In better understanding the habitat requirements of species it is possible to better replicate geomorphic features found on natural rocky shores in engineering designs to improve the amount of refuge (provided on hard structures) from intertidal stressors.

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Chapter 4. Ecological Enhancement of Concrete Tiles- Habitat Complexity at the mm-cm Scale on Vertical Coastal Structures

4.1 Summary

The aim of this study was to produce data to inform how to improve the quality of intertidal habitat on vertical coastal structures such as seawalls. A fine-scale ecological enhancement trial (mm-cm scale) was conducted using 24 cleared natural surfaces and 160 artificial concrete tiles (150x150 mm) of eight different designs at three sites across the UK. Habitat complexity was manipulated in each design so that complexity on the tile surface varied from "low" (plain-cast controls) to "intermediate" (e.g. grooved) to "high" (e.g. pits). Within 18 months, tile designs with intermediate levels of complexity (mm-scale) were optimal in increasing barnacle abundance compared to plain-cast tiles. Tiles with microhabitat recesses up to 30 mm deep (i.e. Singapore design) resulted in higher species richness and mobile species abundance than lower complexity designs. In addition, these high complexity tiles were found to have the lowest peak air temperatures and highest humidity of all designs, highlighting their value as mid-upper shore refugia. This suggests that a hybrid design, using a combination of grooves (mm-scale) and pits (cm-scale), would aid in improving the habitat value of existing and future coastal defence structures, as reported in MacArthur *et al*. (2019) which includes some of the research included here.

4.2 Introduction

Globally, the coastal zone faces increasing pressures from both the landward (urbanisation) and seaward (coastal erosion, sea level rise and increased storm impact driven by climate change) with major implications for coastal ecology and society (Jackson and McIlvenny, 2011; Spalding et al., 2014). In response to these pressures there has been a growth in the number and extent of coastal defence structures over recent decades throughout the UK and beyond, particularly affecting intertidal and shallow subtidal zones (Airoldi et al., 2005). This expansion of coastal defence structures has well-documented geomorphological and ecological implications, including the compression of the intertidal and shallow subtidal zones (coastal squeeze), reducing the intertidal habitat available for organisms and the loss of species diversity (Chapman and Underwood, 2011; Jackson and McIlvenny, 2011).

Species assemblages that occur in artificial habitats are notably different and have typically been observed to have lower species diversity, abundances and altered competitive interactions when compared to adjacent natural habitats (Airoldi et al., 2005; Bulleri and Chapman, 2010; Coombes et al., 2015; Lai et al., 2018). A key factor for variation in species assemblages is the physical difference between coastal defences and natural shores, predominantly the design of structures. Artificial structures are typically highly inclined and homogeneous by design, lacking the habitat structural complexity associated with the geodiversity of natural rocky shores such as fine-scale surface roughness (mm-cm) and microhabitats. This results in artificial structures being poor ecological surrogates for natural shores (Firth et al., 2014b).

Recently there has been an increasing focus on incorporating ecological enhancement into engineering designs, whereby designs are modified to optimise ecological gains by improving the quality or quantity of habitat available (Hall et al., 2018). For example, designing structures to incorporate surfaces and textures that mimic the complexity of rocky shores (surface roughness and microhabitat features i.e. pits and grooves), can improve their capacity to host greater diversity and abundances of species. This increases the ecological value of coastal structures by enhancing their multifunctionality (Naylor et al., 2012a).

Previous studies that have examined the effectiveness of small-scale ecological enhancement have identified a number of surface characteristics that influence recruitment and community composition on artificial structures including texture (Coombes et al., 2011; Hills and Thomason, 1998; Menge et al., 2010), complexity (Lapointe and Bourget, 1999; Loke and Todd, 2016) and colour (Lathlean and Minchinton, 2012). Existing surface texture trials at the mm-cm scale found that intermediate (mm-scale) surface roughness resulted in significantly greater barnacle abundances during the initial settlement season, particularly when compared to smooth and high complexity designs (Coombes et al., 2015). This is of importance as barnacles (an early colonising species) contribute to the development of more diverse species assemblages (Coombes et al., 2015) and can moderate surface microclimate (Coombes et al., 2017). Incorporating high (cm-scale) habitat complexity, such as cm-scale pits has been found to result in significantly greater macroinvertebrate abundances and more diverse communities than smooth surfaces (Firth et al., 2014b; Moschella et al., 2005;

Prendergast et al., 2009). However, the magnitude of these effects are moderated by site-specific factors including local climate and larval supply (Moschella et al., 2005; Sherrard et al., 2016).

On natural rocky shores, microhabitats such as cracks, crevices, pits and rock pools provide important habitat for marine organisms and are important in community development, offering refuge from waves, thermal and desiccation stress, as well as limiting the impacts of predation, scouring and sedimentation on organisms (Firth et al., 2014b; Prendergast et al., 2009; Rickards and Boulding, 2015). This makes heterogeneity a fundamental component in maintaining intertidal biodiversity (Firth et al., 2012; Kostylev et al., 2005). Where these features are lacking on coastal defences, particularly high in the tidal frame, the enhanced risk of desiccation stress (Cartwright and Williams, 2012) is closely linked to microclimate. Habitat complexity is known to influence local microclimatic conditions, affecting desiccation risk (Meager et al., 2011) and in turn influencing the success of colonising intertidal organisms. Increasing the complexity of coastal defences and improving microhabitat quality and quantity (e.g. increased water retention) influences both microclimate and the physical habitat available for intertidal species, resulting in increased biodiversity (Evans et al., 2015; Loke et al., 2019).

This chapter further develops previous fine-scale (mm-cms) complexity trials in an ecological enhancement study on vertical coastal infrastructure at three sites across the UK. Here, habitat complexity (fine-scale surface roughness and microhabitat availability) was manipulated on experimental tiles from the mm to cm scale on eight tile designs, in addition to clearing tile-sized areas on the structure surfaces, across the three UK sites. This was done with the aim of evaluating which surface textures are best placed to maximise the ecological potential for rocky shore species, as measured by species richness (mobile and algal species) and barnacle cover (%). In addition, to evaluate the differences in microclimatic buffering provided by different designs, temperature and humidity data were recorded during a single low tide event at each site. This allowed an assessment into how different scales of ecological enhancement affect habitat quality in terms of both physical habitat space and microclimate. The following overall research question was addressed: How does species richness and abundance compare between experimental tile designs of different levels of habitat complexity?

For this, the following hypotheses were tested:

1. Barnacle cover differs between designs with different levels of mm-cm scale complexity.
2. Barnacle cover is greater on designs of intermediate complexity.
3. Species richness and mobile species abundance is greatest on designs that incorporate microhabitat features (Singapore tile, Art 2).
4. Microhabitats on tile surfaces will provide a more humid environment with lower temperatures than designs with lower complexity.
5. Greater depth and width of features will positively influence species richness and abundance.

4.3 Selecting positions and structures

4.3.1 Orientation

Coastal infrastructure is often vertical or steeply sloping, in stark contrast to the typically horizontal or gentle gradient of natural rocky shores (Chapman, 2006). The notable change in intertidal gradient alongside the extent of coastal infrastructure expansion, considerably reduces the extent of available intertidal habitat (Chapman, 2006). In consideration of this, vertical coastal structures were selected as the test sites for tile installations, as under predicted levels of sea level rise and the expected retreat of many intertidal species higher into the intertidal zone, the presence of coastal infrastructure will present a physical barrier to adaptation (Bellgrove et al., 2013; Jackson and McIlvenny, 2011; Vaselli et al., 2008a). This will considerably impact species that are unable to adapt or survive on steeply sloping artificial surfaces that in any case present an alien and unsuitable habitat.

Vertical structures were selected as they present the greatest opportunities for ecological gains. Mimicking features of high ecological importance on natural rocky shores (rock pools, pits and crevices) (Chapman and Blockley, 2009; Chapman and Bulleri, 2003), are also easily deployed on vertical structures. These retrofitted microhabitats offer cooler and more humid microclimates (Jackson, 2010) than occur on smooth vertical structures. In a review of existing ecological engineering studies, the majority (67% of 109 studies) looked at retrofitting onto existing structures, with only 23% focused on adding or manipulating texture on surfaces (Strain et al., 2017b). In exposed locations retrofitting is deemed unsuitable on engineering grounds (damage and loss) and so in this study only moderately exposed sites were selected. In addition, textured formliners (liners to prepare designs on concrete walls) were used here on vertical coastal infrastructure to offer an alternative form of enhancement that can be used in a variety of exposure settings.

4.3.2 Height within the tidal zone

Experimental test tiles were positioned in the mid-upper intertidal zone at each site (approximately 2 to 3.06 m above mean lower low water (MLLW)). The mid-upper zone is higher within the tidal frame and selected for installation since this part of the intertidal zone is disproportionately lost and affected by the construction of coastal infrastructure (Dugan et al., 2008). In addition, although many studies of this type of enhancement examine the mid intertidal zone, few studies take place at this mid-upper boundary (Figure A 4-1). The loss of habitat in the upper zone reduces the diversity of habitat types available and reduces the potential for the shoreward migration of species and habitat zones predicted under sea level rise (Dugan and Hubbard, 2010; Jackson and McIlvenny, 2011). At mid and upper shore levels especially, species assemblages have been shown to be more homogeneous on artificial coastal structures, supporting fewer intertidal species than at comparable heights on natural rocky shores (Dugan *et al.,* 2008; Lam *et al.*, 2009; Green *et al.*, 2012). Increasing the availability and range of habitat types will result in a greater species diversity and the expansion of their intertidal range (McGuinness and Underwood, 1986; Thompson *et al.*, 1996; Browne and Chapman, 2011).

4.4 Site selection

Experimental tiles were installed on three vertical artificial structures at Blackness Castle in the Firth of Forth, Saltcoats, North Ayrshire and Shanklin, Isle of Wight (IOW) (Figure 4-1). These sites were selected for the reasons outlined in Table 4-1.

Table 4-1. Site Selection Criteria.

Physical/Geographical Parameters	
Tidal Characteristics	The tidal range demarcated coastal areas that were favourable for installation and influenced the installation strategy.
Vertical and Horizontal Extent of Site	Structures had to have a large enough area to install multiple arrays of 14-16 tiles and accommodate spacing to ensure the installation strategy was experimentally sound.
Presence of Hard Coastal Infrastructure	As the tiles were to provide an evidence base to prime further research on coastal bioprotection and enhancement, they needed to be deployed on existing coastal infrastructure to determine the extent of 'enhancement' compared to the existing structure.
Limited Complexity	The tiles had to be deployed on structures that were relatively smooth so that they could be attached securely to the structure.
Ecological Parameters	
Presence of Natural Intertidal Communities	Sites observed to have a dense cover of barnacles and other intertidal organisms.
Social Parameters	
Degree of Anthropogenic Disturbance	The tiles were deployed in areas with relatively low footfall to reduce the risk of disturbance.
Accessibility	Sites had to be accessible by walking with transport nearby for the installation and subsequent monitoring of the tiles.
Permissions	The ownership of land and associated regulations were determined before the installation of the tiles. Permission was successfully granted from the required parties - North Ayrshire Council, Historic Environment Scotland and Artecology.

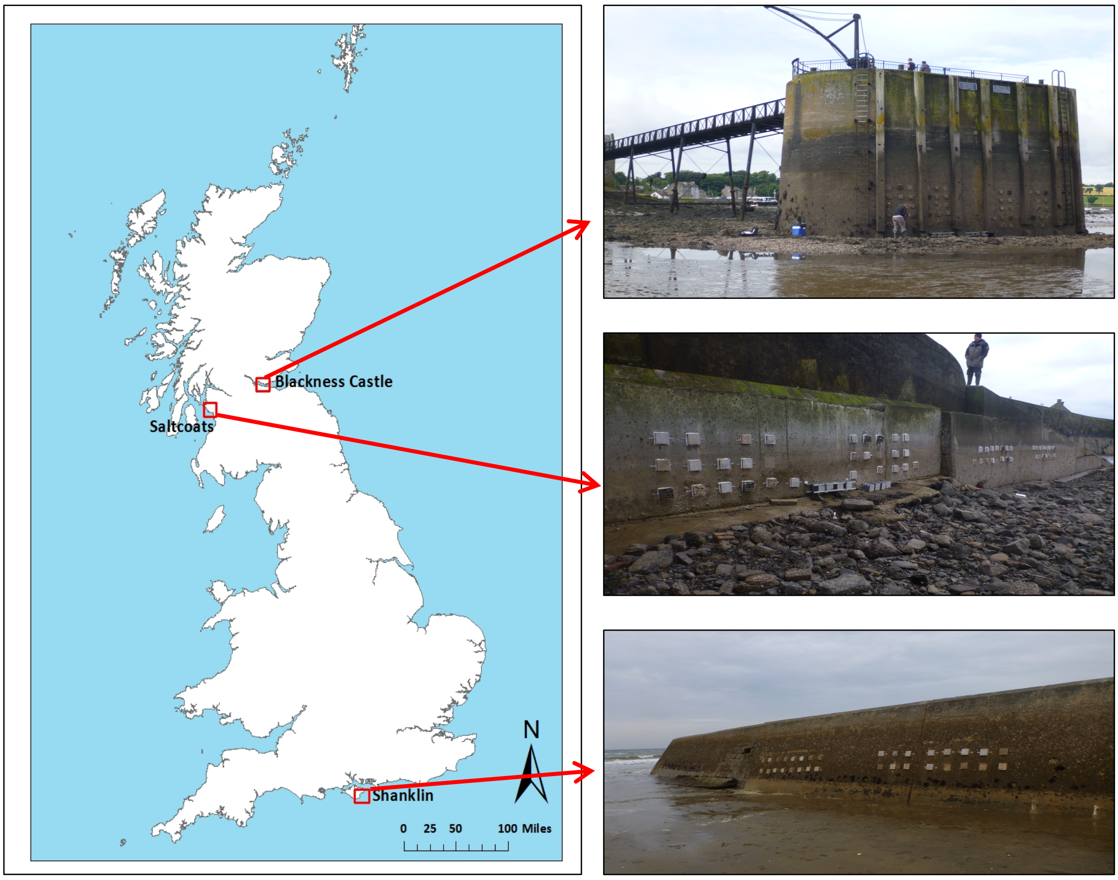

Figure 4-1. Location of tile installations (MacArthur et al., 2019).

4.5 Study Sites

Sites were selected from locations with comparable environmental variables and with similar aspects of the structures where tiles were installed. For each site, background data can be found in Table 4-2. Wave data for Saltcoats and Blackness highlights that the exposure conditions are similar between sites, with both sites moderately exposed (Figure 4-2). Wave data was not collected on the Isle of Wight, but it is estimated that with the location of the structure on the beach, it would approximate the exposure conditions of the Scottish sites.

4.5.1 Blackness

Tiles were installed on the late 19th century concrete pier, attached to the 15th century Blackness Castle, that sits on the southern shore of the Firth of Forth estuary in Scotland. The concrete pier was segmented by wooden beams that resulted in each panel being approximately 1.2 m ±0.1 m wide (Figure 4-1). The surrounding intertidal area is composed of silt and clay, with the lithology of the adjacent natural rocky shore a mixture of sedimentary rock from the mid Mississippian-late Mississippian sub period and olivine analcime-microgabbro from the early Mississippian- mid Pennsylvanian sub period (© Geological Map Data BGS © NERC 2018).

4.5.2 Saltcoats

The seawall at Saltcoats used for installation is divided into two sections. The first section, which has been recently constructed (within the last 5 years), extends 1.83 m vertically and 7.6 m horizontally and the second section of the structure has a height of 2.35 m and a length of 25 m. The lithology within the harbour is mainly composed of sediments of coal measures from the early Pennsylvanian epoch. Sand and gravel deposits are exposed at low tide alongside outcrops of microgabbro bedrock (igneous intrusions) of unknown age (© Geological Map Data BGS © NERC 2018).

4.5.3 Shanklin, Isle of Wight

At Shanklin, the beach plays host to a series of coastal defences including rock armour backed by a high seawall and concrete groynes that play a substantial role in controlling Shanklin Esplanade beach. The concrete groyne selected as an installation site at Shanklin (Figure 4-1) is perpendicular to the surrounding sandy beach and composed of a series of large concrete panels that measure 6 m each, there were six of these panels, measuring 36 m in length with a height of 2.8 m. Geological maps indicate that clay, silt, sand and gravel form the beach and tidal flat deposits, with no notable nearby rocky shore (© Geological Map Data BGS © NERC 2018).

Table 4-2. Background data for each site. Salinity, temperature (°C) and pH values were averaged from n=3 readings.

Site	Coordinates	Structure	Aspect	Salinity (ppt)	Temp. (°C)	pH	Background data collected
Blackness Castle	56°0’24.4614”N, 3°30’55.9362”W	Concrete Pier	NNE	31.20	17.73	8.23	05/08/2016
Saltcoats Harbour	55°37’51.4”N, 4°47’13.4”W	Concrete Seawall	NW	30.70	14.07	8.13	08/08/2016
Shanklin, Isle of Wight	50°38’00.0”N, 1°10’08.5”W	Concrete Groyne	NE	33.87	19.60	7.90	16/09/2016

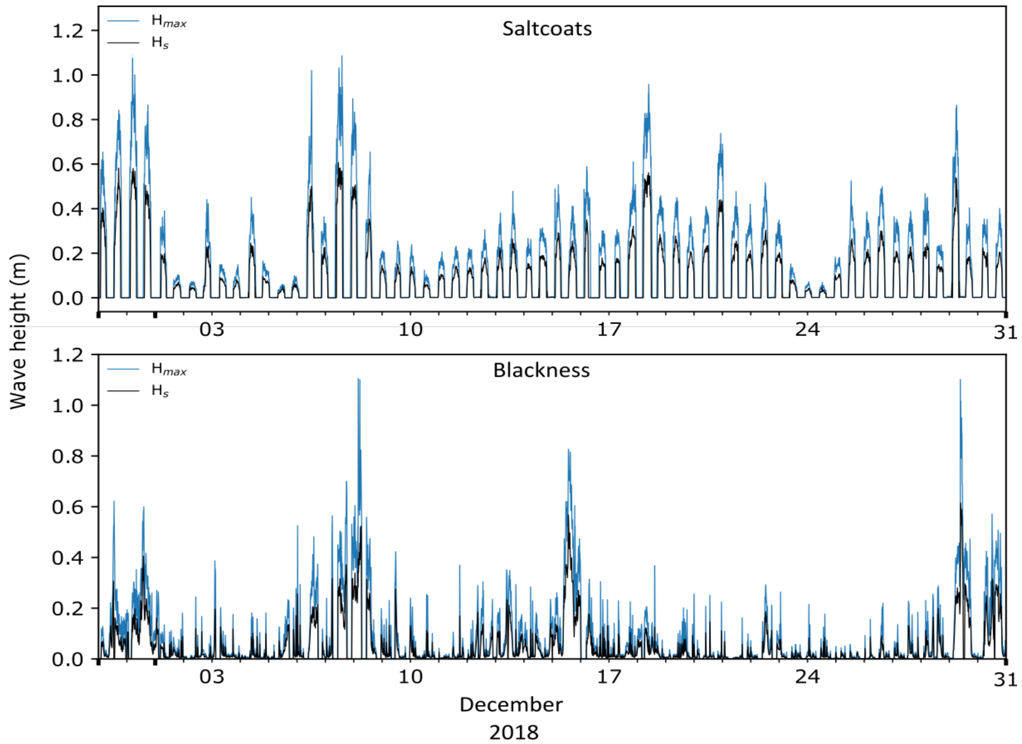

Figure 4-2. Wave data from November 31st- December 31st, 2018 at Blackness and Saltcoats showing max wave height (H*max*) and significant wave height (H*s*).

4.6 Barnacles: key study organisms

Barnacles (Order: Sessilia) were selected as the primary organism for this study due to their global abundance and their notable role as ecosystem engineers with good correlations between the presence of barnacles and mobile taxa abundance (Harley, 2006; Yakovis et al., 2007). Evidence indicates that barnacles are a primary driver of community structure and ecosystem function, leading to the development of more diverse species assemblages (Harley, 2006; Coombes et al., 2015). Exposure to wave energy, heat and desiccation stress are commonplace stressors for intertidal organisms (Dayton, 1971) and even the presence of empty barnacle shells helps buffer these conditions by providing a cooler and wetter microclimate and is a more sheltered habitat than adjacent exposed surfaces (Harley, 2006; Sueiro et al., 2011). Barnacle presence and their provision of biogenic habitat have been observed to increase species richness and abundance when compared to areas lacking this habitat (Thompson et al., 1996). For example, it facilitates the recruitment and survival of *Fucus* spp. (Hawkins, 1981).

Their function extends beyond the provision of habitat and barnacles have been used as indicator species for climate change (Hawkins et al., 2008; Mieszkowska et al., 2006). They play a bioprotective role in limiting the influence of processes such as surface erosion and weathering, either directly or indirectly (Naylor, 2005). For instance, barnacle presence can reduce the impact of weathering and erosion through bioprotection, reducing peak subsurface temperatures and salt ingress (Coombes et al., 2017), likely reducing associated maintenance costs for the structure.

Three species of barnacles were recorded at the sites surveyed, *Semibalanus balanoides* and *Chthamalus montagui,* both native species, and *Austrominius modestus,* a non-native species. Both Saltcoats and the Isle of Wight had all three present, although *Chthamalus montagui* occurred in low abundance at the mid-shore height at Saltcoats. Only *Semibalanus balanoides* and *Austrominius modestus* were recorded at Blackness as this site is beyond the geographical range of *Chthamalus montagui* (Crisp *et al.*, 1981). Newly settled barnacle cyprids are small and do not allow consistent species identification on tiles, so a distinction between these three barnacle species was not made for this study (MacArthur et al., 2019). Breeding patterns vary for each species, with *Semibalanus balanoides* arriving from April through to the end of May, *Autrominius modestus* from May to October and *Chthamalus montagui* starting in August in the English channel and September in the Clyde (Saltcoats) (MacArthur et al., 2019). There is no functional difference between the three species as they are all suspension feeders although minor differences occur in their habitat preferences. *Autrominius modestus* prefers wave sheltered, high suspension load environments, *Chthamalus montagui* is typically found on open coasts and *Semibalanus balanoides* is ubiquitous but rare in extreme wave conditions.

4.7 Tile dimensions and replicates

A review of 61 existing ecological enhancement studies using tiles (also called panels or plates) (Table A 4-1) was undertaken in May 2016 to identify an optimum number of replicates and the ideal size of ecological enhancement tiles for this study (Table 4-3). 150x150 mm was decided as the optimal tile size to account for the outer 30 mm of each tile being excluded for potential edge effects (after Bulleri, 2005).

Table 4-3. Mean tile size and replicates from ecological tile studies review (n=61). See Table A 4-1 for full details

Variable	Mean value (± SE)	Size used in this study
Width	129.86 mm (± 8.39)	150 mm
Depth	124.02 mm (± 7.50)	150 mm
Replicates	7.76 (± 0.57)	N=8 per texture

4.8 Creating and designing the experimental tiles

4.8.1 Marine concrete casting

Five of the eight ecologically enhanced tile designs (150x150 mm) ("Control, "Grooved", "Barnacle", "Geotile" and "Singapore", Section 4.9) were developed in line with engineering standards of practice using a standard marine concrete mix to ensure the durability of the concrete upon exposure to the marine environment (CIRIA, 2010). Concrete cubes (150 mm^3) created from this mix produced a strong concrete, with a mean compressive strength of 52.5 MPa (tested on n=4 cubes). Fine aggregate and water content were adjusted after the initial few batches to reduce the fine aggregate moisture content (Table 4-4).

The concrete mix was weighed out in three layers, firstly the 20 mm washed gravel aggregate was added to the mixing pan, followed by the 10 mm washed gravel aggregate, the fine aggregate (concrete sand) and finally the cement. The proportion of 10 mm and 20 mm was roughly 1:3 of the 10 mm and 2:3 of the 20 mm aggregate. A maximum aggregate size of 20 mm was used as this is the standard in many offshore structures (Mehta, 2002) and these two sizes were used as a graded aggregate was required. This was then mixed in a Cumflow mixer with a 200-litre capacity. Water was slowly added until the mix was uniform and had the correct workability. Rapid hardening cement was used which resulted in the mix setting rapidly and so it was important to maximise efficiency and cast the specimens as quickly as possible.

Table 4-4. Mix composition and quantities for the experimental tiles casting.

Material	Quantity (Initial two batches)	Adjusted Quantities (Later batches)
Mastercrete Rapid Hardening Cement	24.5 kg	24.5 kg
Fine Aggregate (Concrete Sand)	29.3kg	31.2kg
10 mm Aggregate (Washed Gravel)	18.0kg	18.0kg
20 mm Aggregate (Washed Gravel)	36.3kg	36.3kg
Water	9.8kg	7.9kg
Curing Temperature	20°C ±2°C	

To separate the concrete during the casting process, Recticel Eurothane GP Insulation was cut into 150x150 mm cubic spacers. Spacers were cut to three different depths to accommodate the variation in microtopographic relief associated with the silicon moulds. Both the spacers and the silicon moulds were then placed into cubic steel and plastic framework moulds. Tiles were individually cast in these cubic moulds to create 112 marine concrete tiles of five different designs.

Once the concrete was poured into the moulds, they were placed on a vibrating table to remove air bubbles and consolidate the concrete, increasing the density of the resulting cast and adding strength. Surface voids on the concrete castings were smoothed over and the tiles were prepared for field installation by inserting metal brackets into the moulds and allowed to set. The specimens were stored overnight and covered with polythene sheeting to reduce overnight moisture loss. Following demoulding, the tiles were cured in water for 7 days at 20 °C ±2°C to enhance the durability and strength of the concrete by reducing permeability and moisture loss (Chithra and Dhinakaran, 2014).

4.8.2 Art Tiles

Three more designs ("Art 1", "Art 2", "Art 3") were created using Vicat Prompt Natural Cement, with 48 tiles constructed using this material. The mix ingredients for the Art tiles were designed with one part Vicat Prompt Natural Cement to two parts ballast aggregate. Each tile was individually cast, as above, using 700g of the ballast aggregate and 220g of Vicat Prompt Cement, metal brackets were not inserted into these tiles during casting. 48 tiles were constructed using this material. Compressive strength tests (n=4 150 mm^3 cubes) on Art tiles displayed minimal strength of 3.85 mPa. Although this makes the Vicat material tiles unsuitable for use in construction, these designs were included as their ease of casting by hand allows easy integration of the detailed surface features that favour ecology, providing contextual information on optimising designs for ecology.

4.9 Experimental Tile Designs

Designs were based on an understanding of the relationships between complexity (surface roughness and microhabitats i.e. pits and grooves) and biodiversity on rocky shores, drawing from biogeomorphology (i.e. the two-way interactions between organisms and their habitat) and marine ecology concepts. Apart from the Singapore and Art 1 designs, all designs aimed to replicate or integrate topographically complex features of natural rocky shores that are known to influence species recruitment or community composition (Coombes et al., 2011). Further information on each design is found in the sections below.

Although the width and height of each tile was uniform (150x150 mm), depth varied with the casting process and relief of the designs. Depth was calculated by taking an average of n=3 tiles of each design (Table 4-5). Each tile had one textured face to allow for comparisons in the colonisation of the tiles at each site. Designs varied in their complexity (and surface area), providing a gradient from "Low" (plain-cast/clearing), to "Intermediate" (mm-scale modifications to surface roughness), to "High" complexity (microhabitats provided or with areas of relief >10 mm) (Table 4-5) (MacArthur et al., 2019). Designs are hereafter referred to as having low, intermediate or high complexity, with the effects of complexity not disentangled from area in this work.

Wooden panels divided the structure at Blackness into 1 m wide sections, creating spatial restrictions that interfered with the adopted 1.5x tile length separation distance installation strategy. As a result, only n=7 designs installed at Blackness. For other sites, 24 replicates were made of each design (Table 4-5). However, financial restrictions meant that it was not possible to have 24 replicates of each design (n=8 per site) and so some designs were installed at only two sites (n=16 replicates made, Table 4-5).

Table 4-5. Tile replicates, complexity, installation sites and dimensions (mm). Depth taken from n=3 tiles (± SE)

Design	Complexity	Replicates	Sites	Dimensions (mm)
Control	Low	24	All	150x150x32 (± 0 SE)
Grooved	Intermediate	24	All	150x150x25 (± 1.45 SE)
Barnacle	Intermediate	24	All	150x150x27 (± 0.33 SE)
Art 3	Intermediate	16	Saltcoats, IOW	150x150x31 (± 0.58 SE)
Geotile	High	24	All	150x150x52 (± 0 SE)
Singapore	High	16	Saltcoats, Blackness	150x150x55 (± 0 SE)
Art 1	High	16	Blackness, IOW	150x150x34 (± 0.88 SE)
Art 2	High	16	Saltcoats, IOW	150x150x41 (± 0.88 SE)

4.9.1 Cleared surfaces and controls

In this study, two controls were created. The first control strategy was to clear patches of 150x150 mm on the structures used for installation (Figure 4-4A). Disturbance is a natural component of ecosystems. Biological disturbance such as intraspecific competition, grazing and predation (Dayton, 1971; Menge, 1976) and physical disturbance events such as dislodgement from wave exposure, mass mortalities associated with desiccation stress (Benedetti-Cecchi et al., 2000; Denny, 2006), sedimentation and abrasion (Vaselli et al., 2008b) can create bare spaces on substrate for new colonists. These clearings allow monitoring of colonisation on the structure that would occur naturally following a disturbance event. However, these clearings may have been previously weathered and colonised and so are suboptimal controls. Alongside controlled clearings, a plain-cast smooth control tile was attached to replicate a typical pre-cast concrete surface, seen in Figure 4-4B (Coombes et al., 2011; Moschella et al., 2005). The experimental design tiles, the plain-cast tile alongside the natural cleared surface on vertical coastal structures allows for a comparison of the colonisation between the previously uncolonised materials and the baseline that would occur naturally on the structures.

4.9.2 Grooved

A rapidly manufactured design with similarities to fine-scale cracks on natural rocky shores that has been previously shown to significantly increase barnacle settlement compared to plain-cast, smooth (low complexity) and exposed aggregate (high complexity) designs (Coombes et al., 2015). The texture of the original grooved tile was created by dragging a coarse wire brush across semi-dry concrete during the curing process. This resulted in an intermediate level of complexity with a series of regular millimetre scale ridges in the concrete. The design is developed here by making it more replicable at a scale that could be deployed in pre-cast concrete units for coastal infrastructure. The original design was scanned and the groove height increased to make a silicon framework mould for use in the casting process (Figure 4-4C).

4.9.3 Barnacle

This design was created at the University of Falmouth using digital manufacturing equipment (Metcalfe, 2015). The habitat requirements of barnacles were taken into consideration, with this design providing mm-scale relief that scaled up the profile of settled barnacles, including grooved surfaces of different orientations (Figure 4-4D). For this experiment, the original design was scaled up to meet the 150x150 mm specifications of this study.

4.9.4 Geotile

The review in Chapter 1 highlighted that features like crevices on rocky shores are often overlooked. Of the studies that examined these topographic features, many recorded higher species diversity and abundances of examined intertidal species in crevices than in adjacent areas of lower topographic complexity (Healy, 1996; Judge et al., 2009; Scrosati and Heaven, 2008). This is stated to be a result of a number of factors including the potential of these topographic features to buffer fluctuating environmental conditions (Bulleri et al., 2005; Halpin et al., 2002; Kostylev et al., 2005), sustain a moist microclimate and offer refuge (Cartwright and Williams, 2012; Healy, 1996). These features allow organisms to extend their vertical distribution further than they would otherwise on exposed rock (Menge, 1976). With the construction of coastal infrastructure in the mid-upper intertidal zone, replicating these features should be beneficial for intertidal species.

Utilising this understanding, it was determined that replicating the topographic complexity of cracks and crevices could host greater biodiversity potential for the ecologically enhanced design of this tile. The rock surface selected was located in the upper intertidal zone where there would be limited biological growth in order to get a better quality scan of the cracks and crevices (Figure 4-3A). The rock surface was scanned using a TLS Leica C10 with a vertical and horizontal resolution of 1 mm at 1 m distance. This resolution captured the mm-scale microtopography on the rock surface, important since microtopography influences species distribution on rocky shores (Schembri et al., 2005).

Three point clouds were created from the laser scan data of the rock surface (total c. 10 million points at a Multi-Hue intensity value of 0.0785 [min] to 0.4691[max]) and processed in Leica Cyclone 9.0 using target to target registration to a Mean Absolute Error of 1 mm. The point clouds were then cleaned, merged and cropped to a sample area of approximately 1x1 m. The point clouds were further cleaned to 1654092 points and exported to MeshLab_64bit v1.3.4BETA (MeshLab, 2016) from which the deliverable was created (Figure 4-3B). A simplified surface mesh was created using Poisson Surface Reconstruction (Kazhdan et al., 2006), the details of this mesh can be found in Table A 4-2. The depth of features within this design were then exaggerated so that this would test from the mm-cm scale of enhancement (Figure 4-4E).

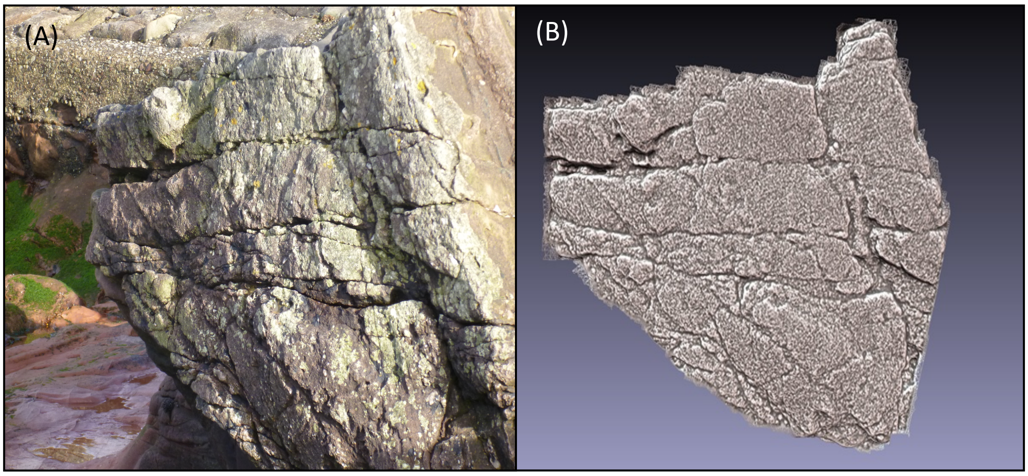

Figure 4-3.(A) Rock surface that was scanned with the TLS. (B) Final product of the mesh that was used to create the 150x150 mm Geotile.

4.9.5 Singapore

The Singapore tile design was previously tested in tropical Singapore (Loke and Todd, 2016) and was included in here to see how a highly complex design (recesses up to 30 mm) would perform in a temperate climate. The design was provided by Lynette Loke and created using the software Complexity for Artificial Substrates (CASU) (Loke et al., 2014). CASU allows for the generation of varying degrees of artificial habitat complexity using different structural components, such as the pits and holes seen in Figure 4-4F.

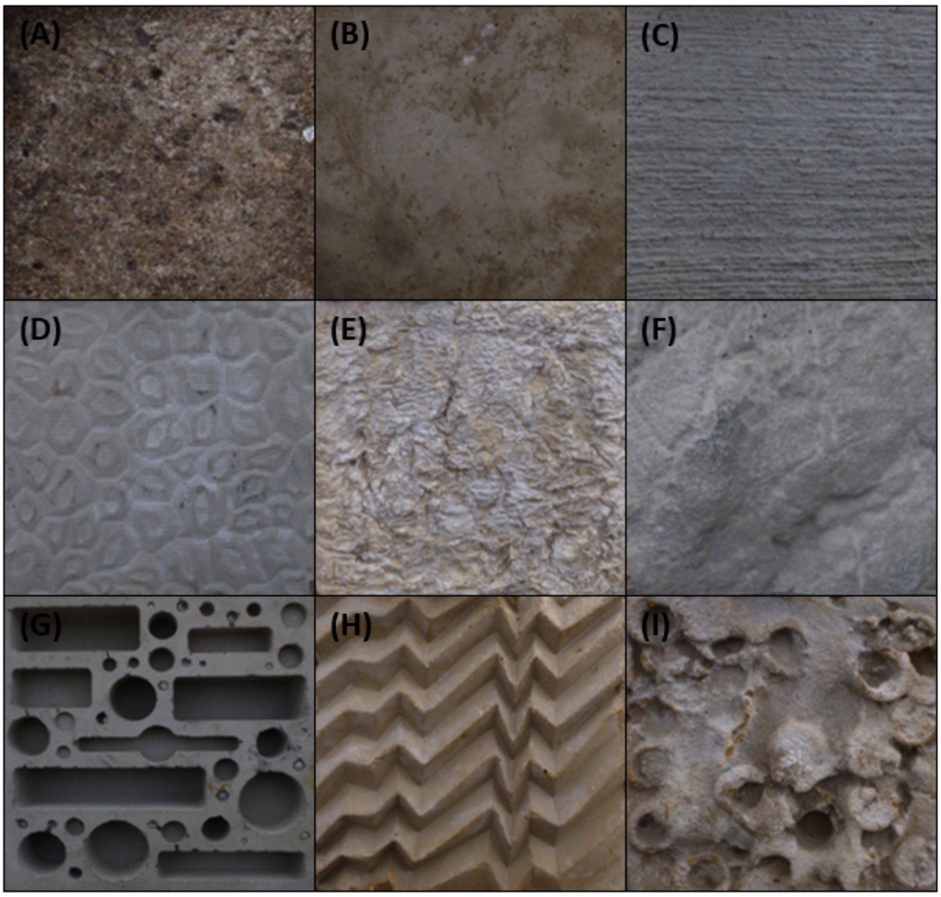

Figure 4-4. Images have been cropped to show the 150x150 mm textured surfaces of a low complexity (A) natural clearing area and (B) Control tile, intermediate complexity (C) Grooved, (D) Barnacle and (E) Art 3 designs and high complexity (F) Geotile, (G) Singapore, (H) Art 1 and (I) Art 2 designs (MacArthur et al., 2019).

4.9.6 Art Tiles

Each of the Art designs aimed to mix art with ecology and create an innovative new habitat and encourage public artistic engagement. The designs were created by Artecology (Artecology, 2019). Three designs were created for this project using everyday materials to form the moulds (Figure 4-4G, H, I). Art 1 was designed by creating a ‘Chevron’ pattern using silicon moulded from folded paper. Art 2 was created by pushing a trowel handle into the sand; the sand-cast design was then used to create a silicon mould. The silicon mould for Art 3 was designed from a crushed-foil cast original.

4.10 Methods

4.10.1 Surface roughness

The textured face of n=1 of each design was scanned using a NextEngine 3D scanner ULTRA HD (accuracy of ±100 micrometres), with no replicates due to the speed of installation needed to catch barnacle settlement after the tiles were made. Tiles were scanned at 8 cm distance and the surface interpolated from points, removing edges to increase accuracy, using an inverse distance weighted technique which created a Digital Elevation Model surface in ArcGIS version 10.6.1. Profile graphs were then created by digitising a 3D line using the interpolate line tool using the 3D Analyst toolbar, with horizontal and vertical profiles generated across the inner 120 mm of each design. For the Art 3 design, only a section of the tile that had broken off was available for scanning but this is still comparable to other designs. Surface roughness was calculated using standard deviation of slope due to its simplicity and accuracy in analysing the whole surface area and ability to examine roughness across multiple scales (Grohmann et al., 2010). Roughness was calculated at the 50x50 micrometres (2500μm^2), 0.5x0.5 millimetres (0.25 mm^2), 1x1 millimetres (1 mm^2) and 5x5 millimetres (25 mm^2) scales to capture the variation in roughness with each design. 25 mm^2 was selected for the maximum scale as there was limited variation between pixel values and this is an optimal size for the size of the scan areas (150x150 mm^2). Mean standard deviations for each scale were then calculated, which allowed for a representative value of surface roughness for the tile faces.

4.10.2 Microhabitat analysis- optimal feature dimensions

The width and depth of each of the microhabitats (“pits”) (mm) on the surface of the Singapore tile was measured prior to installation and noted for subsequent analysis on which feature widths and depths are optimal for mobile species abundance and species richness. This will aid in informing future ecologically enhanced engineering designs.

4.10.3 Baseline surveys

On the initial scoping visit to each site, baseline surveys were conducted within the mid-upper intertidal zone (approximately between 2 and 3.06 m above mean lower low water (MLLW)) along each structure at each site and three vertical transects were set-up at three different horizontal positions. A distance of at least 5-10 m between vertical transects was followed where possible (Chapman and Bulleri, 2003; Petraitis, 1991; Underwood, 1981). At Blackness, where the length of the structure was less than 8 m, quadrat surveys were conducted at 2 m intervals. Although some studies conduct quadrat surveys over a greater horizontal extent (Lagos et al., 2005; Martins et al., 2010), it was determined that the assemblage differences between transects would be limited due to the spatially restricted and homogeneous nature of the structures (Chapman and Bulleri, 2003).

At each vertical transect, n=4 (250x250 mm) quadrats were randomly placed within the intertidal zone, resulting in n=12 quadrats at each site. Surveys were visually conducted using standard non-destructive sampling methods. This provided an insight into baseline barnacle cover (%), species richness (mobile and algal, excluding barnacles) and mobile species abundance. Organisms were identified to species level and barnacles were grouped for percentage cover.

4.10.4 Field installation

Between late April and early May 2016, tiles were installed within the mid-upper intertidal zone at each site to catch the first barnacle settlement of the year (Fish and Fish, 2011). The installation strategy was consistent across the three sites, with structures divided into four separate sections and a spacing of 1-2 metres between sections to allow an even distribution of designs across the structure. Large surface cracks or joints between concrete panels were avoided to ensure that a suitably flat surface was used for tile attachment.

In each section, two replicates of each design were installed, resulting in a total of eight replicates of up to eight designs (including natural clearings) per site. As previously stated, spatial restrictions at Blackness and the cost of casting limited the ability to install all nine designs at each site (Table 4-5). Each section held 14-16 tiles, with tile attachment points delineated and 22.5 cm maintained between tiles in each section (1.5x tile length). These areas were scraped clear of visible organisms using a flat trowel and a coarse wire brush and left for one tidal cycle to be washed. Following this, all cleared areas were burned with a butane blowtorch to ensure no remaining biochemical cues from pre-existing biofilm (Downes and Street, 2005; Green and Crowe, 2013). This allowed all designs and clearings to start from a common baseline in terms of recruitment and larval colonisation (Coombes et al., 2015).

A semi-randomised design was used, with each attachment point assigned a number and designs selected in rotation using a random number generator, to maximise randomisation (Coombes et al., 2015; Herbert and Hawkins, 2006). Tiles were attached directly to the structure using metal brackets, with the textured surface oriented outwards. Holes were drilled into the structures to a depth of 30 or 40 mm, depending on the type of bracket. Stainless steel screws were screwed into plastic Rawl® plugs placed through the holes in the tile brackets, securing the tiles to the structure (Figure 4-5). For additional security, Plastic Padding Marine Filler (marine epoxy) was applied to the edges of each tile prior to affixing the tiles to the structures. Once the tiles were attached to the vertical structure, EVO-STIK 'STICKS LIKE SH*T' was used to fill any gaps between the tiles and the substrate and smoothed to facilitate the movement of grazing species (Herbert and Hawkins, 2006).

Figure 4-5. Tile installation method.

The textured surface of each tile was photographed *in situ* at 2,6,12 and 18 months post-installation to analyse barnacle percentage cover (%), mobile species abundance and species richness of mobile and algal species, excluding barnacles (counted from the presence/absence of species). Only macrofauna and macroflora abundance and richness was analysed in this study. This data was analysed using ImageJ (Schneider et al., 2012) and photographs of the tiles were cropped to the internal 120x120 mm section, with the external 30 mm margin excluded from analyses to account for potential edge effects (after Bulleri, 2005). Barnacle cover was calculated by delineating barnacles (or empty space where barnacle abundance was high) and using the measure function to convert cover to a percentage of tile area (MacArthur et al., 2019). For species richness (mobile and algae, excluding barnacles) and mobile species abundance, the count tool was used across the whole surface of the cropped tile.

4.10.5 Humidity and temperature

To identify whether the presence of microhabitats provide a more suitable microclimate than exposed surfaces (i.e. lower temperature and higher humidity), relative humidity loggers (Hygrochrons™) were attached to tile surfaces over the course of a single low-tide event on a dry, sunny day in summer 2016. Due to the limited number of Hygrochrons available, loggers were fixed using adhesive putty to the centre of n=1 of each tile design, with an additional logger placed in one of the Singapore tile microhabitats and where possible in the crevices on the Geotile and Art 2 tiles. Loggers were deployed with the receding tide and collected as the tide reached the bottom of the structure. The Hygrochrons recorded humidity and temperature at 1 minute intervals over the course of two-three hours (n=120 readings at Blackness and the Isle of Wight, n=180 at Saltcoats) with ±0.6% relative humidity resolution (Coombes et al., 2013b) and ±0.5°C resolution for temperature readings. There was an initial period at the start of each dataset where Hygrochrons were visibly acclimatising to their environment, this was excluded from analysis (after Coombes et al., 2013b).

4.10.6 Statistical Analyses

All statistical analyses were carried out in R version 3.5.1 (R Development Core Team 2018). Generalised least squared (GLS) and linear mixed effect (LME) models were used to analyse the continuous variables of barnacle cover (%), mobile species abundance and mobile/algal species richness. These were tested with respect to the categorical variables of months after installation and tile type to determine the differences in richness and abundance was influenced by the tile designs. Species richness calculations excluded barnacles as these were present on 93.34% of all tiles over the 2-18 month surveys and it was desirable to determine the habitat value of each design for other species in addition to barnacles.

Each site was analysed separately due to very large site differences in baseline barnacle abundance. GLS and LME models were compared using the anova () function in R (Pinheiro and Bates, 2009) , with fixed effects kept the same for comparison and the model with the lowest Akaike Information Criterion (AIC) was selected as being the most suitable for the data (Zuur et al., 2009). Models selected for each site and ecological variable are listed in Table A 4-3 . For post-hoc pairwise comparisons between tiles types within each month after installation, least-squares means were conducted after applying Tukey's multiplicity adjustment (Lenth, 2016), with all tests performed at the 95% confidence level. Full statistical results table for barnacle cover, species richness and mobile abundance are available in Table A 4-4, Table A 4-5 and Table A 4-6 respectively. Kruskal-Wallis tests were used to determine differences in surface roughness, temperature and relative humidity (%) data between tile designs at each site.

4.11 Results

4.11.1 Surface roughness

As no replicate scans were undertaken due to time constraints with the installation, grouping tiles across scales of roughness (n=4), indicated that there was a significant difference between designs ($H[7]=15.216$, $p=0.0333$) but further post-hoc testing was not significant.

Standard deviation of slope increased across all designs as scale increased (Figure 4-6). The Singapore tile had the greatest surface roughness at all the examined scales and the standard deviation of the results was highly variable with this design (Figure 4-6) due to the variation in feature depth combined with the smooth surface. The other "high" complexity designs (Geotile, Art 1, Art 2) varied in which had the greatest roughness depending on the examined scale. The Geotile was best at 25 mm², Art 1 at 2500µm² and Art 2 at 0.25 mm² and 1 mm² (Figure 4-6). Designs of intermediate complexity varied most from the smooth controls at the 2500µm² and 0.25 mm² scales, with the difference between the barnacle and control tiles lowering as the scale increased whilst the grooved design consistently had greater roughness than the control across all the examined scales (Figure 4-6).

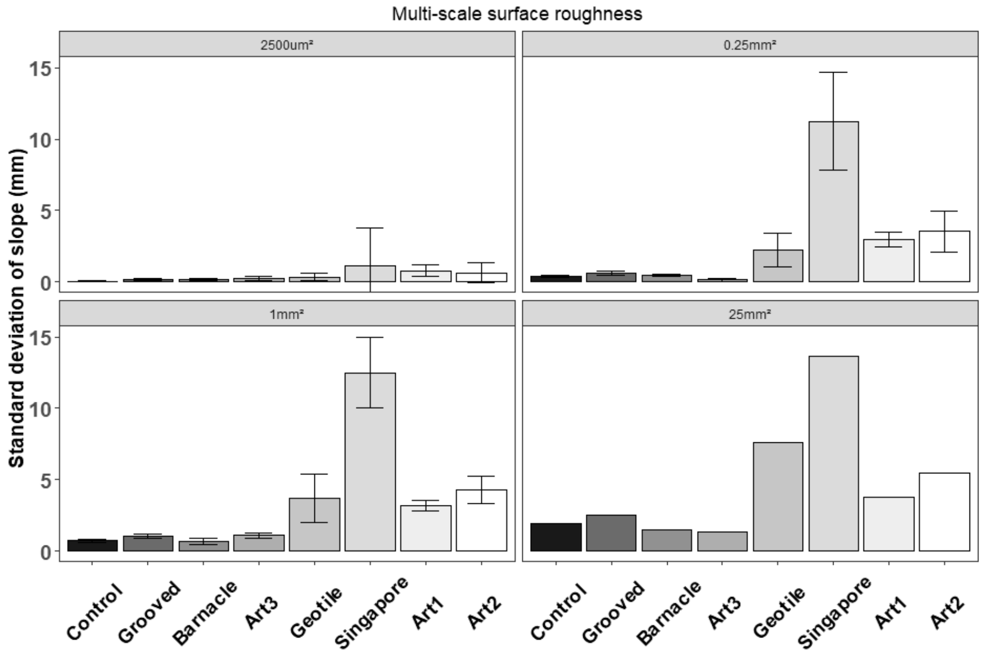

Figure 4-6. Surface roughness, calculated from standard deviation of slope, of the textured surfaces of ecologically enhanced tiles at 2500µm² (50x50 µm), 0.25 mm² (0.5x0.5 mm), 1 mm² (1x1 mm) and 25 mm² (5x5 mm) scales. Error bars show the standard deviation of results. Final scale (25 mm^2) had standard deviations <0.01 of pixels.

Figure 4-7 shows the surface roughness profiles in relief (mm) of each design. Scans taken of the control, grooved and barnacle designs were not level and so appear sloping when they are flat surfaces. However, the variation in roughness can be identified, with the surfaces of the intermediate complexity designs (Art 3, Grooved, Barnacle) optimally varied at the mm-scale to incorporate features (such as grooves) optimal for the body size of settling barnacles (approximately 0.5 mm) (Coombes et al., 2015). These profiles additionally show the variation in the high complexity designs, with the Singapore and Geotile tiles having around 30 mm of relief and the varied depth in microhabitats that the Singapore tile provides.

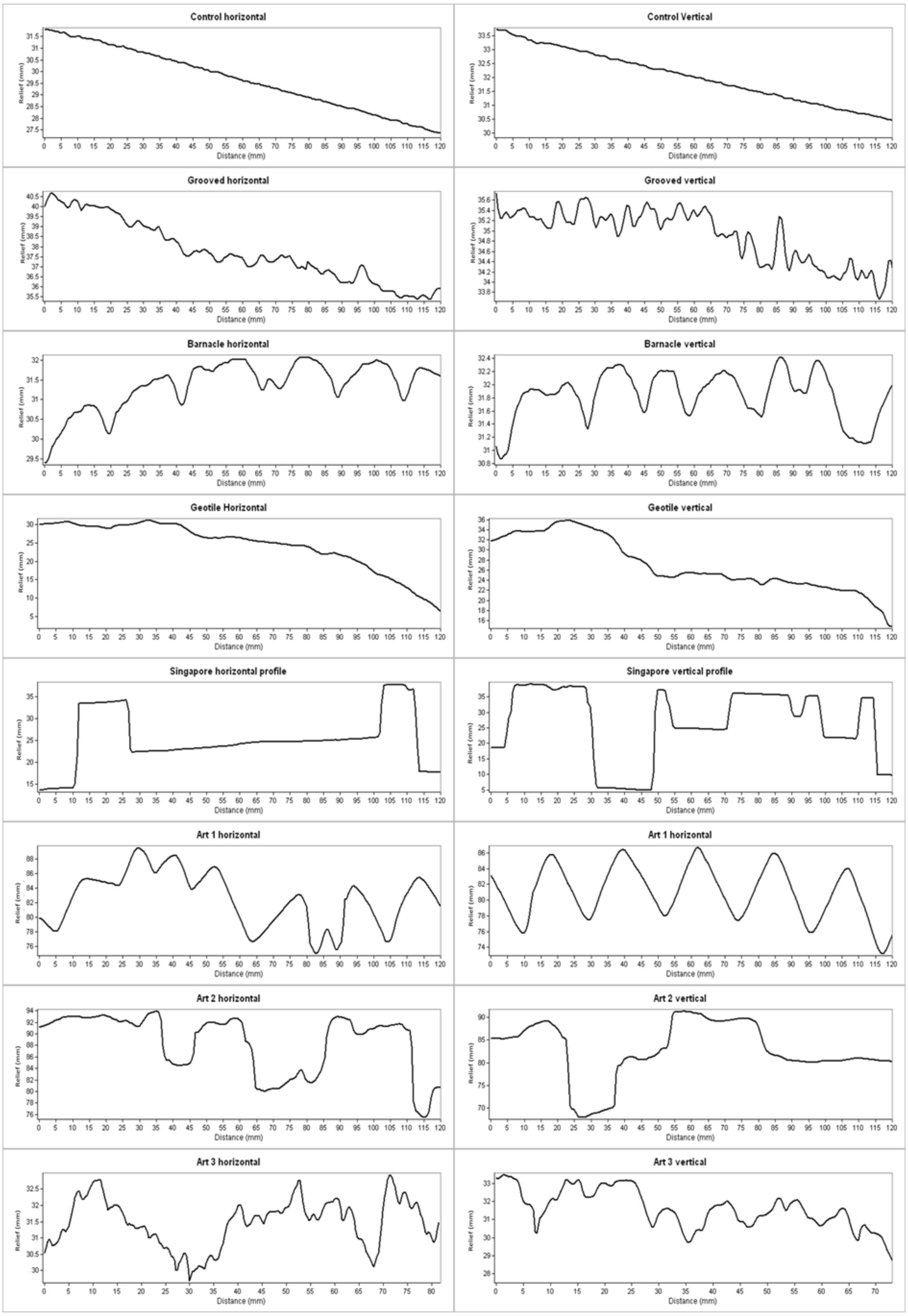

Figure 4-7. Surface profiles of experimental tile designs (horizontal and vertical profiles) with distance along the tile (mm) and relief (mm).

4.11.2 Baseline surveys

Barnacle cover varied considerably between sites (n=12 quadrats per site). At Blackness, barnacles occupied an average of 90.42% (±1.56 SE) of available space, a value that was consistent across the height of the installed tiles, possibly resulting from its more sheltered estuarine location. Saltcoats had an average of 34.33% (±9.02 SE) barnacle cover possibly reflecting the young age of the seawall as half of the installation area was constructed within a couple of years prior to installation. The Isle of Wight, a more open coast site, averaged 53.33% (±8.36 SE) barnacle cover across the structure.

Species richness counts, excluding barnacles, indicated that each of the examined sites was relatively species-poor. The average abundance (recorded in brackets ± standard error) of each of these species per quadrat was relatively low at each site. Three species were recorded on the Isle of Wight (*Littorina littorea* (0.5± 0.23 SE), *Patella vulgata* (9±1.91 SE) and *Mytilus edulis* (0.08 ±0.08 SE) and Blackness (*Littorina littorea* (5.58± 1.76 SE), *Littorina saxatilis* (0.33 ± 0.19 SE) and *Patella vulgata* (3.83 ± 0.73 SE) and four species at Saltcoats (*Littorina littorea* (15 ± 2.75 SE), *Littorina saxatilis* (0.08 ±0.08 SE), *Patella vulgata* (1.75 ± 0.68 SE) and *Nucella lapillus* (0.08 ±0.08 SE). No algal species were recorded in these surveys. The seawall at Saltcoats had high densities of *Littorina littorea* as these were mostly observed inside the shells of dead barnacles and in surface cracks on the seawall.

4.11.3 Barnacle abundance

Barnacle cover (%) was compared between tile designs within each site. Plain-cast control designs were found to underperform in terms of both early recruitment and long-term cover of barnacles compared to the textured tile designs at all sites, particularly those with intermediate complexity (Grooved, Barnacle, Art 3). For example, across all sites and months after installation, the Grooved design outperformed control tiles ($p<0.05$). The Barnacle and Art 3 designs also exceeded the cover of control tiles across all months on the Isle of Wight ($p<0.001$), with the same pattern observed between Barnacle and control designs at Blackness ($p<0.05$) and the Art 3 design and the control from 2-12 months at Saltcoats ($p<0.001$). Control tiles also did not match clearings, which were rapidly colonised at all sites to equal most designs in their ability to attract barnacles (Figure 4-8).

Several of the high complexity designs (i.e. Art 1, Geotile and Singapore) attracted fewer barnacles than designs with intermediate complexity (Figure 4-8). The grooved design significantly outperformed the Art 1 and Singapore designs where they were installed (Saltcoats and the Isle of Wight for Art 1, Blackness and Saltcoats for Singapore) across all monitoring periods ($p<0.01$). The Barnacle design was also found to have a greater number of barnacles than the Art 1 and Singapore designs at Blackness across all months post-installation ($p<0.001$). On the Isle of Wight, this was also true of the Barnacle versus the Art 1 design from 6 to 18 months ($p<0.05$) and the Art 3 compared to the Art 1 design at all months ($p<0.01$). In examining the Geotile, the grooved design was a more suitable settlement surface for barnacles across all sampling periods on the Isle of Wight ($p<0.01$), up to 6 months post-installation at Saltcoats ($p<0.05$) and up to 12 months at Blackness ($p<0.01$). The Barnacle design also exceeded the Geotile on the Isle of Wight at 2 and 6 months ($p<0.05$) but did not differ at Blackness or Saltcoats. Art 3 was also more suitable than the Geotile at Saltcoats (2-6 months, $p<0.01$) and on the Isle of Wight (2-12 months, $p<0.01$).

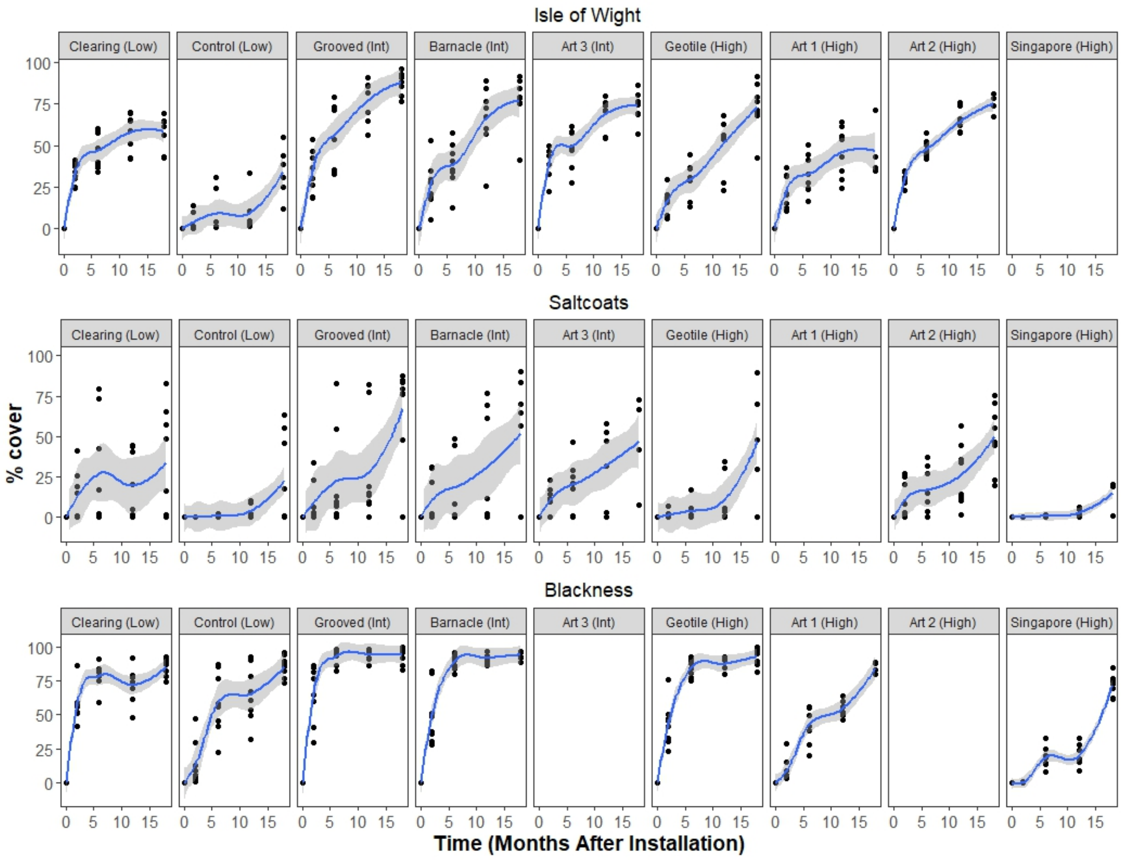

Figure 4-8. Barnacle percentage cover (%) with months after installation (2, 6, 12 and 18 months) at the Isle of Wight, Saltcoats and Blackness (installed between April and May 2016). Panels show designs and complexity levels indicated in brackets (L=low, I=intermediate and H=high). Grey band width indicates 95% confidence interval. Blank boxes indicate designs not installed at that site (MacArthur et al., 2019).

The Singapore design was the most complex, with a high number of microhabitats of varying depths. This was less suitable for barnacles than clearings at Saltcoats up to 6 months post-installation ($p<0.01$) and across all sampling stages at Blackness up to 18 months post-installation ($t(196)=5.078$, $p<0.001$). Barnacle cover was also significantly lower on the Singapore tile than the control tile at Blackness from 6-18 months ($p<0.01$) and did not differ from the control tile at Saltcoats. In contrast, the Art 2 design matched designs of intermediate complexity (Grooved, Barnacle, Art 3) at Saltcoats and the Isle of Wight where it was installed and outperformed the Singapore and control tiles across all sampling periods at Saltcoats ($p<0.001$) and control tiles on the Isle of Wight ($p<0.001$).

Of all the designs examined, the intermediate complexity of the grooved tile was found to be optimal for rapid colonisation and a high percentage cover of barnacles (Figure 4-9). Although there was variation in which intermediate complexity design (Art 3, Grooved, Barnacle) performed the best at each site and with each monitoring visit, the grooved tile outperformed the barnacle design at Saltcoats and Blackness, although the results are not statistically significant. Additionally, on the Isle of Wight the grooved tiles were found to have significantly greater barnacle cover than the barnacle design at 2, 6 and 12 months ($p<0.05$) and Art 3, exclusively at 18 months, ($t(219)=-3.103$, $p<0.05$). These findings validate hypothesis 1 and 2, that barnacle cover would vary with design and that intermediate complexity would promote the greatest levels of barnacle cover.

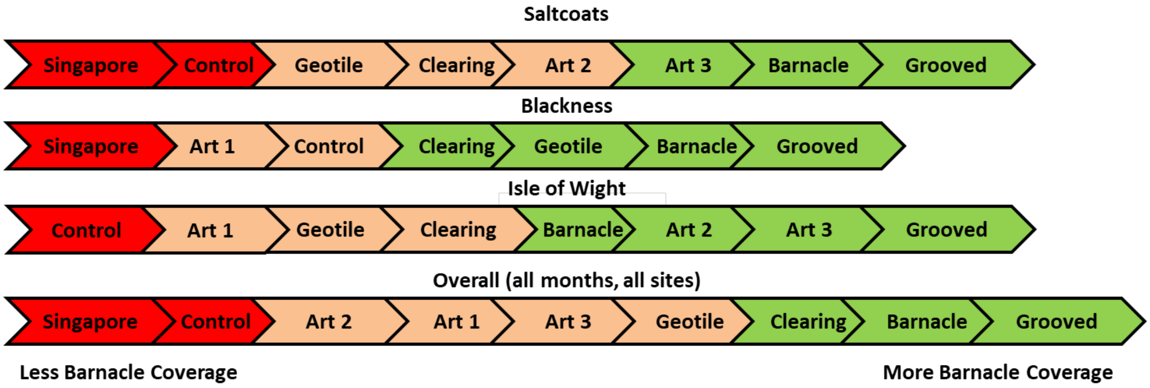

Figure 4-9. Schematic of optimal designs for barnacle colonisation at each site from least (left) to most (right) optimal with red indicating poor performance, orange for intermediate and green for the best performing designs. Calculated from mean barnacle cover (%) on each tile type at each site across all months (2, 6, 12 and 18). For overall, sites have been additionally grouped.

4.11.4 Species richness and abundance

In comparing the influence of design on intertidal species, a low, intermediate and high complexity design (Control, Grooved and Singapore/Art 2 respectively) were compared against the baseline in Table 4-6. Barnacle species were excluded for this as they were present across all sites and species richness and abundance counts are used to evaluate habitat provision for other species besides barnacles. The Singapore design matched baseline values for the total number of species within 2 months at Blackness and 6 months at Saltcoats (Table 4-6) and increased the frequency of occurrence of species, particularly at Saltcoats where n=8 species were observed at 18 months. In contrast, the low and intermediate complexity designs did not match baseline species numbers apart from the 18 month data on the control tiles at Saltcoats (n=5). More species were observed in the microhabitats on the Singapore tile over the 18 month monitoring period at Saltcoats and Blackness (9 and 5 taxa respectively) than on the grooved designs and control designs (Table 4-6). The Art 2 design was a substitute comparison for the Singapore on the IOW as this design was not installed here. Observations showed that it performed moderately better than the grooved design in the early stages of colonisation (2 and 6 months) but otherwise matched the intermediate complexity design and did not exceed the baseline. Most species were observed on each of the three tile designs but *Littorina obtusata, Gibbula umbilicalus, Carcinus maenas* and *Fucus spiralis* were unique to the pits of the Singapore tile (Table 4-6) and as such, the Singapore tile is the best performing design in terms of attracting the greatest number of species.

Table 4-6. Presence of species with each monitoring period for low (control), intermediate (grooved) and high (Singapore/Art 2) complexity tiles. B= baseline, C= control, G= grooved, S=Singapore, A2 =Art 2 where Singapore was absent from installation. ● (2 months)○ (6 months) ■ (12 months) □ (18 months) ✓(baseline).

	Saltcoats				Blackness				Isle of Wight			
Species	**B**	**C**	**G**	**S**	**B**	**C**	**G**	**S**	**B**	**C**	**G**	**A2**
Patella vulgata	✓			■	✓	□	○	■ □	✓	● ○ ■ □	●○ ■□	●○ ■□
Littorina littorea	✓	●□	●○ ■□	●○ ■□	✓	○ ■ □	○ ■ □	● ○ ■ □	✓	○ □	■□	●○ ■□
Littorina saxatilis	✓	□		○■ □	✓	●■ □	○ ■ □	● ○ ■ □			○ □	○ □
Littorina obtusata				○■ □								
Gibbula umbilicalis				●○ □								
Melarhaphe neritoides		□	□	■□								
Nucella lapillus	✓	□	□	□				□				

Carcinus maenas								● ○				
Mytilus edulis									✓			
Ulva sp.		●○ ■□	●■	□						●	●	●
Fucus spiralis				□								
Total Species	**4**	**5**	**4**	**9**	**3**	**3**	**3**	**5**	**3**	**3**	**4**	**4**
Baseline	4				3				3			
2 months		2	2	2		1	0	3		2	2	3
6 months		1	1	4		1	3	3		2	2	3
12 months		1	2	5		2	2	3		1	2	2
18 months		5	3	8		3	2	4		2	3	3

The greatest abundance and largest number of species observed (excluding barnacles) was recorded at 18 months post-installation on a Singapore tile at Saltcoats, with 27 counts of n=5 species in the 120x120 mm examined area (11 *Melarhaphe neritoides*, 14 *Littorina littorea*, one *Littorina saxatilis,* one *Littorina obtusata* and 29.78% *Ulva* sp. cover) (MacArthur et al., 2019). Of the intermediate complexity designs, there was a maximum of 3 species recorded across all months at all sites. The species found on the tiles were typically from the *Gastropoda* class, with the most commonly observed species being *Littorina littorea, Littorina saxatilis* and *Patella vulgata* and low abundances of *Littorina obtusata, Gibbula umbilicalis, Nucella lapillus* and *Melarhaphe neritoides* recorded. One individual *Carcinus maenas* was additionally recorded in one of the microhabitats of the Singapore tile at Blackness at 2 months and another at 6 months. In examining algal species, ephemeral green algae (*Ulva* sp.) was present in varying abundances and covered most tiles at 2 months post-installation. Small amounts of *Fucus spiralis* had also started colonising a microhabitat on an Art 2 tile at Saltcoats at 2 months (0.36% cover) and on two Art 2 tiles at 12 months ($\bar{x}$=0.80%). At 18 months, *Fucus spiralis* was observed inside microhabitat features on Singapore tiles at Saltcoats ($\bar{x}$=3.34%, n=2) and on one Art 2 tile (0.21%).

The Singapore tile was the best performing design in terms of both species richness and abundance at the sites where it was installed (Figure 4-10 and Figure 4-11). At Saltcoats, the Singapore tile had significantly greater species richness and abundance than several designs of lower complexity across the 2-18 month monitoring period. From 6-18 months, species richness was higher on the Singapore tile than clearings, Control, Grooved, Barnacle, Geotile and Art 3 designs ($p<0.001$) and additionally on the Art 2 design at 12 and 18 months ($p<0.001$). The Singapore design attracted consistently greater numbers of mobile species than the Control, Grooved, Barnacle and Geotile designs throughout the period of monitoring ($p<0.001$) and Art 3 from 2-12 months ($p<0.05$). Clearings attracted high mobile species abundance at Saltcoats but were significantly less successful than the Singapore design at 6 and 12 months ($p<0.05$), with other months having no significant difference between the two. The Art 2 design also performed well at Saltcoats, having higher species richness than the Barnacle, Control and Geotile designs at 2, 6 and 12 months post-installation ($p<0.05$), the Art 3 design at 6 months ($t(215)=3.357$, $p<0.05$) and the grooved design at 6 and 12 months ($p<0.01$). Additionally, the Art 2 design outperformed the Control, Barnacle, Geotile and Grooved designs in attracting mobile species across all monitoring visits ($p<0.01$) and the Art 3 design at 6 months ($t(215)=3.156$, $p<0.05$).

At Blackness, the Singapore tile had significantly greater species richness and mobile abundance than all the tile designs between 2 and 6 months ($p<0.05$ for Control, Grooved, Barnacle, Art 1, Geotile). Species richness continued to be greater on the Singapore tile than all designs, including clearings, at 12 months ($p<0.05$), with the Barnacle and Geotile designs less successful at attracting species, even after 18 months ($p<0.001$). Mobile abundance continued to be greater on the Singapore design than all others, apart from the Grooved, at 12 months ($p<0.05$) and the Barnacle tile remained less successful at attracting mobile species up to 18 months ($t(203)=-3.998$, $p<0.01$).

Intermediate complexity designs did not perform well in terms of species richness and abundance compared to high complexity tiles (Figure 4-10, Figure 4-11). However, the intermediate Grooved design at Blackness had a greater mobile abundance than the Barnacle designs at 12 and 18 months ($p<0.001$), with an average of 2.06 individuals over 12 and 18 months compared to 1.88. As more barnacle shells became empty at this stage, more mobile species were being found inside empty barnacle shells at the site.

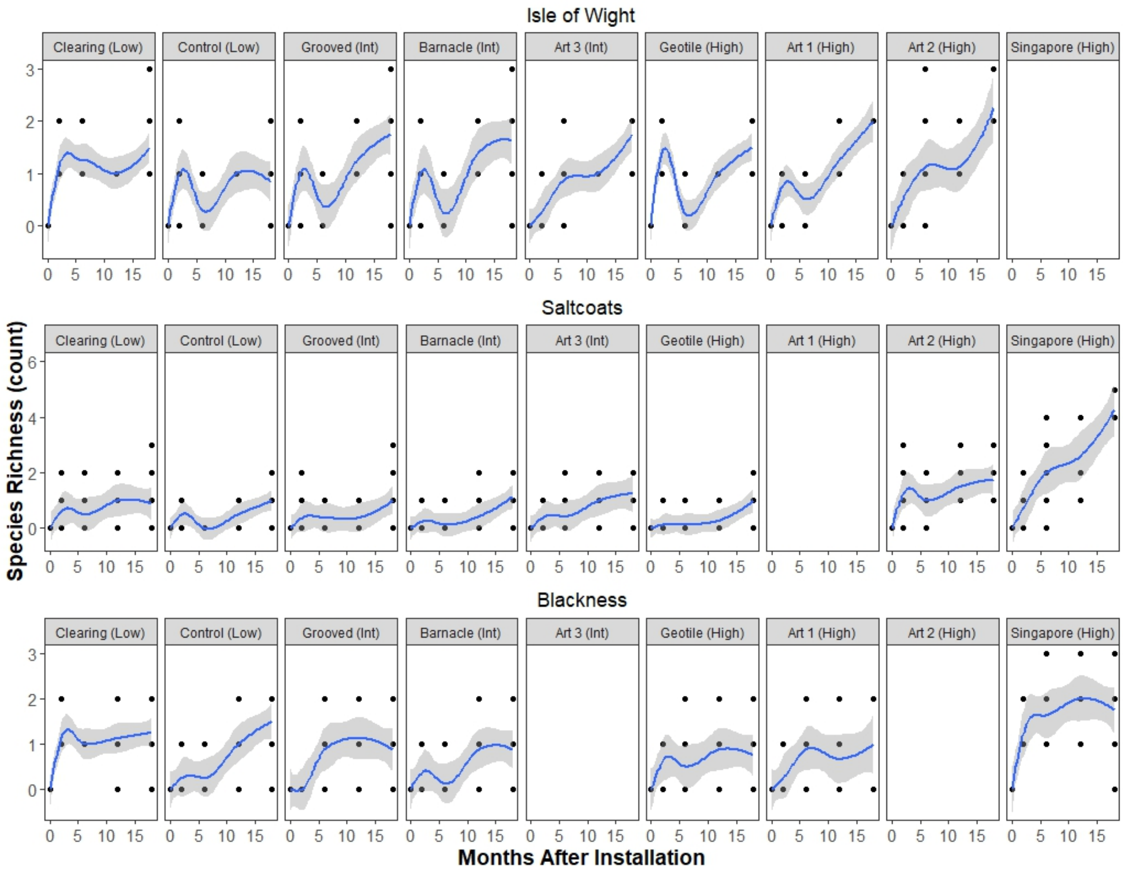

Figure 4-10. Species richness over time (2, 6, 12 and 18 months) at the Isle of Wight, Saltcoats and Blackness (installed between April and May 2016). Panels show designs and complexity levels indicated in brackets (L=low, I=intermediate and H=high). Grey band width indicates 95% confidence interval. Blank boxes indicate no installation at that site.

On the Isle of Wight, there was limited differences between tile designs for species richness (Figure 4-10), but the Art 1, Art 2 and Art 3 designs performed better than the control tiles at 18 months ($p<0.05$) and all designs exceeded the control in attracting mobile species at 12 and 18 months post-installation ($p<0.01$).

Clearings had greater species richness than several designs (Art 1, Barnacle, Control, Grooved) at Blackness at 2 months ($p<0.01$). In terms of mobile abundance, clearings performed well across all sites, with greater abundance than multiple designs at 2 months at Blackness (Control, Barnacle, Geotile, Art 1, Grooved $p<0.001$) due to the high numbers of *Patella vulgata* (2.88 ± 0.69 SE per tile), with this pattern continuing at 6 months (Control, Barnacle, Geotile, Art 1, $p<0.001$) and 12 months (Barnacle, $t(203)=-4.331$, $p<0.001$). On the Isle of Wight, clearings had greater mobile abundance than all designs at 2 and 6 months ($p<0.001$), which had an average of 2.63 and 5.13 species respectively, this was predominately limpets (n=20 and n=38 total at 2 and 6 months). At Saltcoats, clearings were observed to exceed the Barnacle, Control and Geotile designs at 6 and 12 months ($p<0.05$, Figure 4-11).

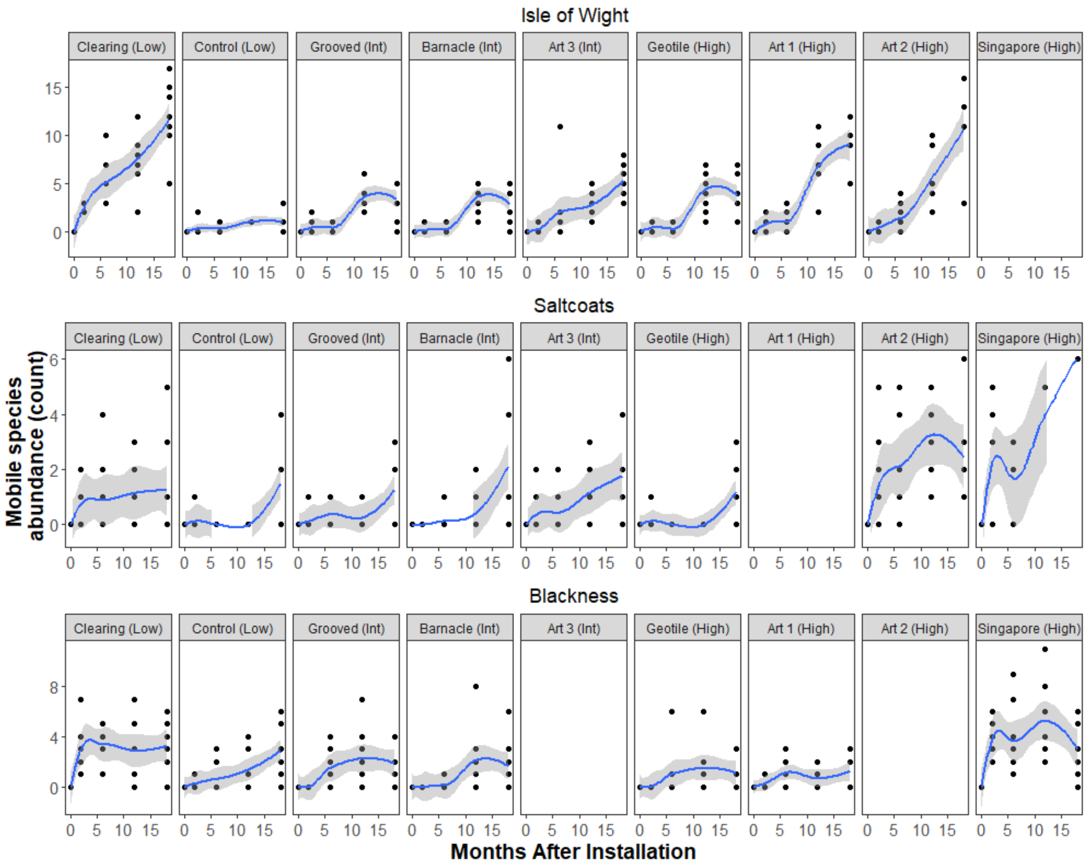

Figure 4-11. Mobile species abundance between 2 and 18 months at the Isle of Wight, Saltcoats and Blackness with designs from low to high levels of complexity. Grey band width indicates 95% confidence interval.

Overall, the best design for species richness and abundance is the Singapore tile, with the Art 2 design also performing well (Figure 4-12). Although the Geotile had high complexity, it did not perform well at any of the examined sites. Tiles of intermediate complexity varied in their ability to attract a range and high abundance of species. The findings indicate that hypothesis 3, that species richness and mobile species abundance is greatest on designs that incorporate microhabitat features, can be accepted, although it is acknowledged that there will be regionally specific variation in how successful designs are in promoting species richness and abundance.

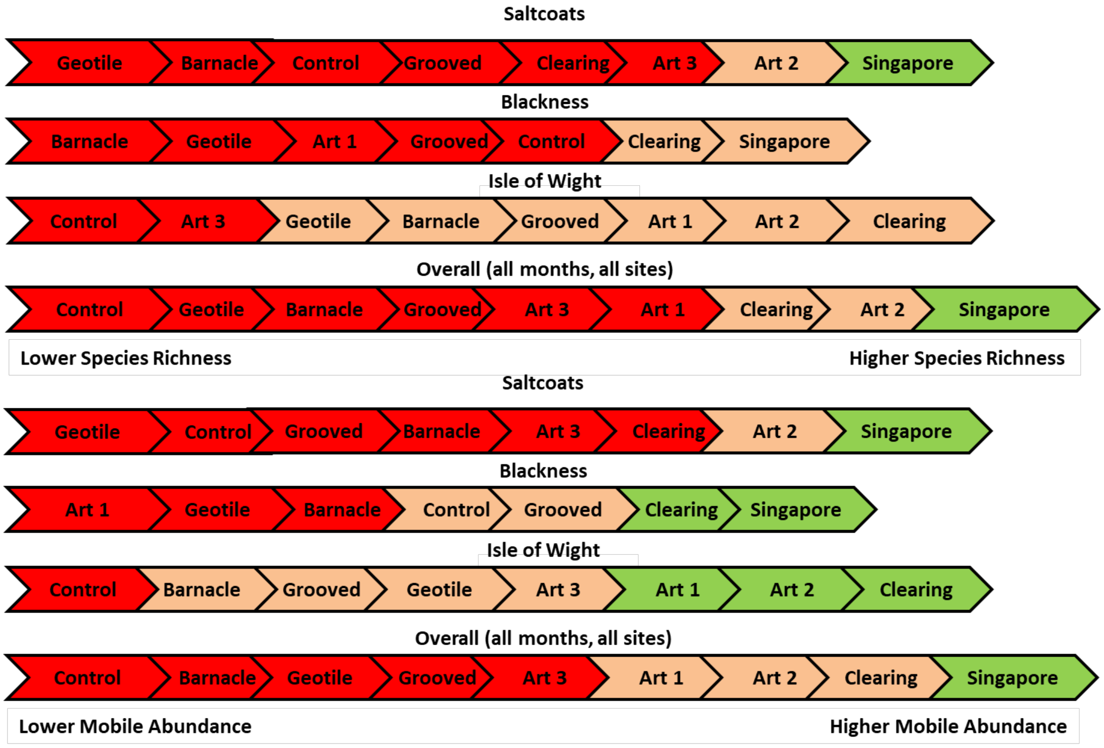

Figure 4-12. Schematic of optimal designs for species richness and mobile species abundance at each site from least (left) to most (right) with red indicating poor ecological performance, orange for intermediate and green for the best performing designs. Calculated from mean species richness (mobile and algae, excluding barnacles) and mobile species abundance on each tile type at each site across all months (2, 6, 12 and 18). For overall, sites have been additionally grouped and averaged.

4.11.5 Humidity and temperature

Humidity and temperature significantly differed between tile types at all sites ($p<0.001$). The incorporation of microhabitats into the designs altered the microclimate at the scale of individual tiles. The Singapore tile had higher humidity and lower temperatures than all tile designs of lower complexity at Saltcoats and Blackness ($p<0.001$). This was also true of the Singapore surface at Saltcoats which had a higher humidity than the Art 2 (surface), Art 3, Geotile (crevice) and grooved designs ($p<0.001$) and lower temperatures than the Art 2 tile (surface and microhabitat) and Art 3 designs (MacArthur et al., 2019). At Saltcoats the Singapore tile averaged 83.38% relative humidity across the microhabitat and surface measurement, 4.85% higher than the next best reading from the Art 2 microhabitat. Peak temperatures were additionally the lowest on the Singapore tile, with the tile surface having the lowest peak temperature of 14.54°C

(0.052°C lower than the microhabitat). At Blackness, the Singapore microhabitat had an average of 94.04% humidity (±1.99 SD) compared to the plain-cast control at 80.04% (±2.67 SD). Variance in humidity was lowest in the Singapore microhabitats (7.36% and 3.96% for Saltcoats and Blackness respectively) and there was low variance in temperature conditions (0.23°C and 0.33°C for Saltcoats and Blackness).

There was also a notable difference between microhabitat and the tile surface, with humidity higher in the microhabitat than on the Singapore tile surface ($p<0.001$) at Saltcoats and Blackness (Figure 4-13). The microhabitat of the Singapore tile had 7.22% (Saltcoats) and 13.13% (Blackness) higher humidity on average than the tile surface (MacArthur et al., 2019).

The Art 2 design, which was the other design to include notable microhabitats, was found to have higher humidity in microhabitats than the surface of the Control, Grooved and Barnacle designs at both the Isle of Wight and Saltcoats ($p<0.001$). The Art 2 design averaged 63.34% humidity on the Isle of Wight and this varied very little over the course of the low tide period examined (4.21% variance). Additionally, the microhabitat had significantly lower temperatures than clearings, Barnacle and Grooved tiles at Saltcoats and the Control, Grooved and Geotile (crevice) tiles on the Isle of Wight ($p<0.001$) (MacArthur et al., 2019), with low variability in temperature across the low tide (0.23°C). On the surface of the Art 2 design at Saltcoats, there was higher humidity (clearing, Barnacle and Control, $p<0.001$) and lower temperatures than several other designs (clearing, Control, Barnacle, Grooved and Geotile (crevice), $p<0.001$).

The best performing design on the Isle of Wight, where the Singapore design was not installed, was the Art 1 design. This had higher humidity and lower temperatures than the Control, Grooved, Barnacle, Geotile (crevice and surface) and Art 3 designs ($p<0.001$) and additionally had lower temperatures than both the surface and microhabitat on the Art 2 design ($p<0.001$), likely due to the high relief (approximately 12 mm). The Art 1 design was only moderately higher in humidity than the Art 2 microhabitat (1.16% difference) and there was very little difference in temperature between the two, with the Art 1 design 0.19°C lower on average. Both the Art 1 and Art 2 designs offer suitable microclimates for species in the absence of more pronounced microhabitats like the Singapore design.

The Control and Grooved tiles were frequently found to have lower humidity and higher temperatures than several other designs at each site with a high amount of variance, particularly at the Isle of Wight and Blackness. The Control tile at the Isle of Wight had the highest variability (7.62% and 0.35°C) whilst the Grooved design was highly variable in its conditions at Blackness and Saltcoats (13.69% and 0.36°C and 9.28% and 0.23°C respectively). The Grooved design was only exceeded in its variability over the low tide by the Geotile crevice at Blackness (14.60% and 0.30°C). These findings allow hypothesis 4, that microhabitats on the tile surface will provide a more humid environment with lower temperatures than designs with lower complexity, to be confirmed and that the lower complexity designs do not provide a suitable microclimate for intertidal species.

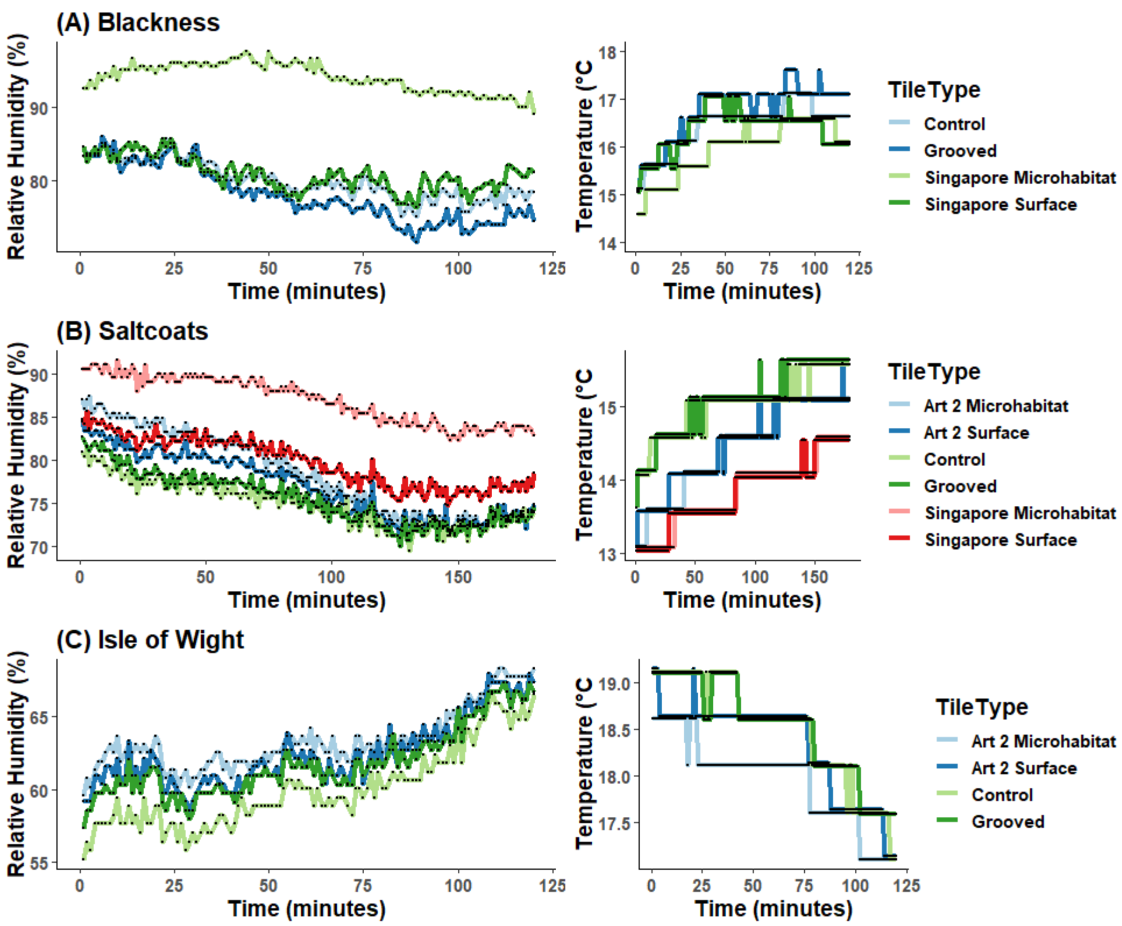

Figure 4-13. Time series data of relative humidity (%) and temperature (°C) on the surface of a low, intermediate and high complexity design at (A) Blackness, (B) Saltcoats and (C) Isle of Wight.

4.11.6 Influence of feature dimensions (width/depth) on species richness and abundance

The Singapore tile was composed of a series of circular and rectangular microhabitats ranging in width and depth (min/max width= 3-89 mm and min/max depth= 3-30 mm, excluding surface). Of the relationships examined at both sites, only feature width (mm) was positively correlated with mobile species abundance at Blackness (r=0.7, p=0.011) (Figure 4-14). No other relationships at either site were found to correlate strongly (Figure 4-14). Hypothesis 5 can therefore be rejected as feature dimensions at this scale do not seem to positively influence species richness and abundance.

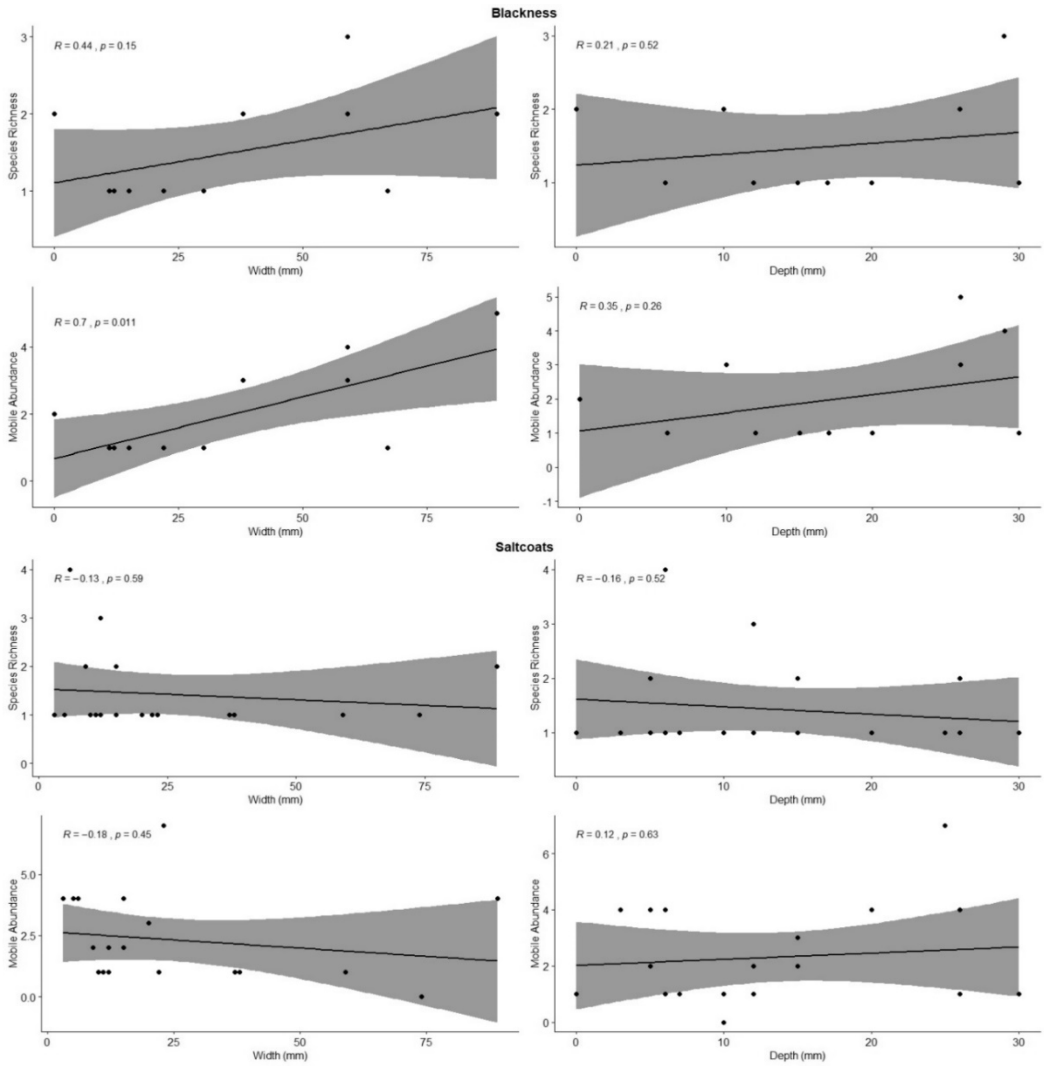

Figure 4-14. Pearson's correlations between species richness and mobile abundance with width and depth of microhabitats on the Singapore tile (mm) at 18 months post-installation at Blackness (n=12) and Saltcoats (n=19). Grey area indicates 95% confidence interval.

4.12 Discussion

At the scale of these enhancements, the addition of intermediate levels of complexity increases the rate and cover of barnacle colonisation and the addition of microhabitats (high complexity) in the form of 'pits' increases both species richness and abundance, particularly when compared to plain-cast controls.

The influence of habitat complexity on barnacle colonisation varied with site, likely a result of differences in local larval supply (Minchinton and Scheibling, 1991). Clearings were found to have rapid rates of barnacle colonisation, often exceeding or equalling the best performing designs during early monitoring (2-6 months). At high densities, barnacle cyprids have been known to favour the availability of free space and be less selective in their habitat preferences (Kent et al., 2003). The gregarious nature of barnacle settlement (Kent et al., 2003) coupled with the high baseline presence of nearby barnacles, particularly at Blackness, would have promoted more rapid colonisation of the cleared areas. This helps explain the pattern of barnacle cover at Blackness, where even control tiles had high barnacle occupancy, with preferred barnacle habitats (mm-scale complexity) filling up faster giving an apparent greater affinity of larvae for less complex tiles (MacArthur et al., 2019). Low baseline abundances of barnacles were recorded at Saltcoats and on the Isle of Wight, which produced a positive density dependency with barnacles more likely to fill their preferred habitat.

Compared to other designs, the smooth concrete of plain-cast control tiles had significantly less barnacle cover than textured designs, particularly compared to intermediate complexity designs (mm-scale modifications), which were optimal in attracting high barnacle cover. Of the three intermediate designs, the Grooved tile was the best at promoting the early recruitment and colonisation of barnacles. The ridges of the grooved tiles had the highest roughness of the intermediate complexity designs, particularly at the 25 mm^2 scale. The original design that this work builds on (Coombes et al., 2015) had grooves ranging from 0.5-2 mm, which was exaggerated here by up to 5 mm and performed successfully in attracting high barnacle cover. These intermediate ridges allowed cyprids to settle in high densities, with the grooves ideally suited to the small body size and attachment methods of barnacle cyprids (Coombes et al., 2015).

Barnacles were observed to uniformly line up across these textured ridges at each site as this design allowed cyprids to settle in high densities with little distance between individuals (Crisp, 1961).

The addition of this mm-scale complexity for barnacles is important in the design of coastal structures as barnacles have been shown to provide a protective layer which performs a bioprotective function for structures by reducing surface damage by heat and salt (Coombes et al., 2017). Barnacles also have ecological engineering potential in providing biogenic habitat through the presence of live and dead barnacles, which provides habitat structures for other organisms (such as littorinids) to settle (Harley, 2006; Thompson et al., 1996) and promotes early community development (Cartwright and Williams, 2012; Harley, 2006) such as encouraging the settlement of Fucus zoospores (Van Tamelen and Stekoll, 1997). Biogenic habitat can also offer refuge from intertidal stressors such as wave action and desiccation stress (Rickards and Boulding, 2015).

The significant differences in barnacle cover between plain-cast and intermediate complexity designs were prominent during the early stages of recruitment from 2-6 months and highlight the need for a textured surface on hard coastal defences, especially where barnacles are prevalent. Promoting the early colonisation of barnacles at a high coverage maximises the potential bioprotective properties and promotes wider ecological and community development benefits. It should be noted that the extent of the success of these designs will be site-dependent and is reliant on both the tidal height of the structures, degree of shelter, the time of year when construction commences and the available larval supply (Minchinton and Scheibling, 1991; Coombes et al., 2015; Jonsson et al., 2004). The success of these interventions will be maximised at sites with low baseline barnacle abundance, such as Saltcoats, as there will be a greater need for complexity to encourage barnacle recruitment to protect structures. In sites with high baseline abundance, such as the estuarine Blackness, saturation of barnacle abundance occurs much faster on intermediate complexity designs whereas other sites may see the effects of complexity over longer time periods. Each site must have detailed baseline surveys conducted to determine the optimal design required for the site.

Higher levels of complexity contribute towards the composition and functioning of ecological communities and the addition of microhabitats at this scale (up to 30 mm deep) significantly increased species richness and abundance compared to controls and several other designs of lower complexity at the sites where the Singapore tile was installed and additionally exceeded baseline species richness at 12 months at Saltcoats. This is in line with previous findings where higher levels of complexity resulted in more diverse communities and greater mobile species abundance (Moschella et al., 2005; Prendergast et al., 2009), even when surface area was subject to experimental control (Loke and Todd, 2016). The variation in the width and the depth of the pits of this tile had little influence on species richness and abundance and for ease of replication, future designs could be more consistent with the size of microhabitats. The complexity of the Singapore design positively altered the tile microclimate producing the highest humidity and lowest temperature of all the designs at the two Scottish sites. This likely contributed to its success in attracting a greater abundance and variety of species as it offers a refuge that buffers the fluctuating environmental conditions and associated stressors (e.g. predation, desiccation) of the mid-upper intertidal zone (Cartwright and Williams, 2012; Kostylev et al., 2005).

Initially, most of the tiles were colonised by the opportunistic *Ulva* sp. and after 18 months, those tiles with microhabitats (Art 2, Singapore) were observed to have fucoid algae (*F. spiralis*) growing within the pits at Saltcoats. This is indicative of typical rocky shore succession patterns (Martins et al., 2007) and suggests that the higher complexity promotes a faster rate of succession than smoother designs. Longer term monitoring is required to determine when the assemblages on the tiles stabilise.

However, the success of the Singapore design is limited by the smooth surface between the microhabitats (i.e. it lacks roughness at the mm-scale). It was the least effective of all the designs, including the plain-cast controls, in promoting barnacle cover. This reduces the ability of barnacles to facilitate further community development and provide important biogenic habitat (Thompson et al., 1996) and highlights that smooth surfaces are not suitable surrogates for the surface roughness typically found in natural rocky shores. In contrast, the designs of intermediate complexity (Grooved, Barnacle, Art 3) had significantly greater barnacle colonisation than designs with lower and higher complexity across all three sites. This is consistent with previous studies whereby higher levels of surface roughness do not necessarily equate to the most favourable scale for early colonising species, such as barnacles, where cyprids have a length around 0.5 mm (Coombes et al., 2015).

The work presented here can ideally be scaled up in future to create full scale ecological formworks rather than the 150x150 mm test tiles; preliminary work to design an ecoformliner based on these results and those of others in the UK, is currently ongoing with the intention of using it as part of a live flood alleviation scheme in Portsmouth.

A few notable improvements could be made to maximise habitat quality for UK intertidal zones in semi-sheltered to exposed areas. One of the features lacking from the enhanced tile designs was the ability to retain water, usually absent from existing coastal defences, and limiting both the potential diversity of colonising species and the ability of desiccation-sensitive species to expand their vertical distribution onto hard coastal structures (Browne and Chapman, 2011; Evans et al., 2015; Firth et al., 2013). The 'Geotile' was designed with the aim of replicating crevice habitat to trap water but this was difficult to replicate effectively. The Singapore tile had the greatest potential to retain water, but the pits were angled 90° from the surface and while this resulted in a more humid microclimate, no water was retained. Angling the interior of the pits 45° downwards would allow water to be trapped and would further improve biodiversity, microhabitat and refuge availability (Evans et al., 2015). The optimal design to maximise species richness, abundance and take advantage of the bioprotective and ecosystem engineering potential of barnacles would be a hybrid between the Singapore design, with cm-scale microhabitats angled to retain water, combined with the intermediate complexity of the grooved design, maximising multi-scale complexity (mm-cm).

4.13 Conclusion

This study has shown that enhancing hard coastal structures by using designs with intermediate scale complexity promotes more rapid barnacle colonisation. This leads to fine-scale biogenic habitat creation and aids the development of more diverse and functional community assemblages. Adding microhabitats between 10-30 mm depth and width, provides habitat that would otherwise be absent on these structures and increases species richness and abundance. Adding microhabitats serves to moderate the microclimate of the tile surface by increasing humidity and decreasing temperatures, this buffers the negative effect of intertidal stressors. This results in increased habitat quality on vertical coastal structures. Ongoing monitoring is key to determine the influence of these design features over the longer timescales that match the design life of the engineering structure proposed (typically ~80-100 years).

In future, optimal designs for ecoformwork for use in UK waters should be created with ecology in mind through identifying multi-scale designs that incorporate mm-scale grooves and cm-scale microhabitats. The creation of pre-cast ecologically enhanced formwork does not impact the engineeering performance or maintenance of the structure but would maximise the quality and quantity of habitat on hard coastal infrastructure. Adding texture to commercially available formwork only increases the cost by around 0.1-0.6% across the whole scheme, which is a negligible cost compared to the costing of the entire sea defence (Naylor et al., 2017a).

Future projects should upscale these enhanced designs but also conduct detailed baselines so that designs are site specific, since each site may perform differently in different contexts and should be judged against engineering specifications and any ecological and biodiversity mitigation requirements that may be demanded by statutory agencies. There is a need for more collaborative ventures between ecologists, geomorphologists and engineers in order to achieve more sustainable and multifunctional hard coastal infrastructure that maximises ecological gains by providing suitable habitat. Urgent attention should be given to identifying multi-scale designs to optimise for algal and mobile species abundance and richness in order to mitigate habitat loss created by the engineering structure itself as well as to partially offset any future habitat loss predicted under climate change.

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Chapter 5. Passive Ecological Enhancement- Optimising Ecological Suitability of Rock Armour Revetments Using Rock Material and Rock Mass Properties: the Hartlepool Headland Coastal Defence Scheme

5.1 Summary

Rock armour revetments deployed as coastal defences lack the surface complexity of natural rocky shores and this reduces their habitat potential. Whilst ecological enhancement trials have created artificial features on coastal structures (such as retrofitting rock pools) to improve ecological suitability, the Hartlepool Headland coastal protection scheme used passive enhancements. This involved selection of the most ecologically suitable lithology (lighter colour, rougher surface) of the options available and, during installation, optimising the positioning of boulders.

Different combinations of rock mass and rock material properties can potentially accelerate early ecological colonisation of rock armour boulders and improve habitat suitability and heterogeneity. However, limited attention has been given in the past to the ecological suitability of the lithologies used in coastal engineering, so this study undertook a series of laboratory tests on several commonly used lithologies to identify which material properties best influence ecological suitability (e.g. surface roughness, porosity, albedo and rock chemistry). These results found that, through careful material choice, a combination of high albedo (light coloured) and high surface roughness optimises ecological value.

Field testing was undertaken at Hartlepool to elaborate on these findings. As the rock armour revetment at Hartlepool was installed in sections over a period of years, a series of monitoring studies aimed to examine how fully passively enhanced Shap granite boulders (recommended for use over Norwegian granite due to its lighter colour and rougher surface), compared to partially enhanced boulders (lithology optimised but not positioned) post installation. An additional field trial compared the Shap granite at Hartlepool with Norwegian granite of similar age (2 years post installation) at nearby Skinningrove. The Shap granite was passively positioned to optimise existing surface features on the boulders, whereas the Norwegian granite at Skinningrove was not passively positioned, allowing a comparison of the effectiveness of this type of enhancement on early colonisation. Early colonising species (barnacles) were found to

favour the lighter Shap granite that had been passively positioned for larger scale features, such as dm-scale ledges, leading to a statistically significant increase in mobile species abundance compared to the non-enhanced Norwegian granite. Such enhancement across the mm-dm scale can maximise ecological gains and both lithology and rock mass properties offers an effective and inexpensive form of passive enhancement. This approach is best suited to high-energy environments where rock armour revetments are the preferred option of coastal defence.

5.2 Introduction

The lithologies commonly used for the construction of hard coastal defences such as boulder revetments are selected for their resistance to abrasion and weathering, functional performance, lifetime and cost (French, 2001; Crossman et al. 2003) and are usually very different in their physical characteristics from the local coastal environment. Rock armour revetment boulders have been favoured in recent years due to their durability, low maintenance costs and availability of suitable quarried product of appropriate sizing and quality (Bradbury et al., 1998; French, 2001). Revetment boulders are freely placed to be structurally stable and protect from scour and erosion by dissipating wave energy with the physical properties of the rock, durability requirements and cost of transport and quarrying all factors in selecting the lithologies used. The designs of these boulder revetments are location specific, with consideration given to erosion risk, water levels and environmental conditions (Crossman et al., 2003).

Hard coastal engineering structures (such as sea walls, rock revetments, breakwaters) are typically built using fresh, unweathered rock, and sometimes concrete, materials that often lack the surface complexity so important for ecology. It is well documented that artificial coastal defence structures have different community compositions and lower species richness and abundance than natural rocky shores (Bulleri and Chapman, 2010; Chapman and Bulleri, 2003; Gacia et al., 2007; Pister, 2009), differences often attributed to reduced surface heterogeneity of these structures (Moschella et al., 2005). Material type (Green et al., 2012), orientation (Glasby and Connell, 2001) and habitat availability also influence community composition and diversity. Age also exerts an influence with many new structures not exposed for long enough to allow weathering processes to increase ecological suitability by modifying the physical properties of rock and marine concrete (Coombes et al., 2013a).

Comparing natural shores with artificial structures, differences in geomorphology and thus habitat availability are evident across a range of physical scales. At the micro-scale (µm-mm), the substratum type and surface roughness of materials influences early stage colonisation and subsequent community development and functioning (Coombes et al., 2011; Green et al., 2012). Carefully selecting material types for surface roughness or long-term biogeomorphological potential (the interplay between geomorphology and species (Naylor et al., 2012b) through bioerosion by boring species and bioweathering by pitting of the rock surface (Coombes et al., 2011)), has the potential to increase the availability of habitat at the micro-scale. At the meso-scale (cm-m), more complex microhabitats such as cracks, crevices, rock pools and ledges offer greater surface area for settlement and higher niche availability for intertidal organisms (Strain et al., 2017b). In addition, these features offer refuge against a range of intertidal environmental and biological stressors, including desiccation stress, wave stress and predation (Bulleri and Chapman, 2010; Gray and Hodgson, 2004; Loke and Todd, 2016).

Where hard coastal infrastructure is favoured over more nature-based solutions, such as managed realignment, there is an increasing body of work promoting the benefits of increased successful colonisation, diversity and abundance of species on artificial structures by increasing their habitat quality. This ‘ecological enhancement’ or ‘ecological engineering’ has developed in innovative ways aiming to increase the complexity and area of enhanced habitat over a range of physical scales. Strategies for ecological enhancement can be usefully separated into two forms: active enhancement and passive enhancement.

Active ecological enhancement attempts to mimic the geomorphological complexity of natural rocky shores. Examples include modifying the chemistry of marine concrete to improve its ecological suitability (Perkol-Finkel and Sella, 2014), mm-cm scale texturing of concrete to promote rapid species colonisation (Coombes et al., 2015; Loke et al., 2014, Chapter 4) and retrofitting existing forms of rock armour and sea walls with holes (Evans et al., 2015; Firth et al., 2014b), pools (Browne and Chapman, 2011; Firth et al., 2014b) and ledges. These active enhancements can be added across multiple physical scales (mm-cm, cm-dm, dm-m).

Passive ecological enhancement attempts to make informed decisions on the choice of lithology and positioning of boulders used in revetments and are often low-cost options that are simple to implement during planning and construction. Careful selection of lithology for rock armour involves the examination of its rock material properties. This involves selecting lithologies that are chemically and/or physically (e.g. light in colour or rough surfaced) optimal for maximising their ecological suitability (Coombes et al., 2011;2013). Passive positioning involves deliberately placing boulders to utilise their natural surface heterogeneity, e.g. surface depressions facing upward to mimic natural rocky shore features such as pools and ledges (Naylor et al., 2017b).

Rocky intertidal species are widely known to prefer topographically complex surfaces that reduce the effects of environmental stressors including temperature (Kordas et al., 2014) and desiccation (McAfee et al., 2016). Microclimate stressors are predicted to increase under a changing climate (Brierley and Kingsford, 2009) and so passive enhancement through the selection of suitable rock material types is of importance for future climate change contexts. Rock material properties such as colour, porosity and fine-scale roughness (Coombes and Naylor, 2012) have been found to moderate some of these environmental stressors and the selection of lighter coloured, chemically favourable (high calcium) soft lithologies with high surface roughness at the µm-cm scale can maximise ecological potential of artificial structures (Coombes et al., 2015; Sempere-Valverde et al., 2018).

Lighter coloured lithologies with higher albedos and lower surface temperature (Coombes et al., 2017; McGreevy, 1985) reduce the thermal stress experienced by colonising species and affects survival and distribution. The influence of rock colour on community development depends on local climatic conditions, higher albedo has advantages for species in a warmer or warming climate with experimental tests on dark backgrounds showing more limited ecological colonisation than light ones (Kordas et al., 2014). More porous lithologies exhibit higher ecological suitability as the surface remains wetter for longer during the stressful low tide period (Coombes and Naylor, 2012). Using rougher material from the onset of construction offers fine scale enhancement, important for the build-up of marine biofilm and community development and encouraging the settlement of early colonising species, such as barnacles (Cacabelos et al., 2016; Chabot and Bourget, 1988; Coombes et al., 2015; Sempere-Valverde et al., 2018). Finally, the chemistry and hardness of lithology can influence the diversity and distribution of intertidal species, with softer and more chemically favourable lithologies offering greater biogeomorphological potential and remaining wetter and more thermally suitable during tidal cycles (Coombes and Naylor, 2012; Naylor et al., 2012b).

Passive positioning involves orienting individual boulders to make use of any pre-existing surface heterogeneity, such as the surface depressions that mimic natural rocky shore features such as pools, pits and ledges, to maximise the ecological suitability of the structure (Naylor et al., 2017b). This maximises available habitat complexity and increases the potential for water-holding capacity of boulder surfaces. Typically, boulders when randomly placed without consideration of positioning lack the range of water-retaining microhabitats that are important in reducing thermal and desiccation stresses that affect the distribution and physiology of rocky intertidal species. Microhabitats tend to maintain lower temperature and higher humidity, offering refuge from wave impact and other stressful abiotic conditions (Lee and Li, 2013; Rickards and Boulding, 2015). On rock armour, where boulders are positioned favourably, surface depressions deep enough to retain water serve to mimic natural rock pools, favouring biodiversity and increasing the presence of key intertidal species. This attribute may help meet any environmental mitigation measures required by planning authorities without compromising on structural integrity and engineering design guidelines.

To determine the range of ecological enhancement studies and identify the gaps in the literature, a systematic review was conducted using Google Scholar and Web of Science for ecological enhancement studies in coastal marine environments. The search string accounted for a variety of enhancement types and underlying structures as well as enhancement studies conducted on natural rocky shores. The following search was used on studies from 2010-2019 to gather the most recent and relevant results, ("ecological enhancement*" OR "ecological engineering") AND ("coastal defence*" OR "rock armour" OR "rock revetment") AND ("microhabitat*" OR "rock pool*" OR "roughness"

OR "texture*" OR "groove*" OR "pit*" OR "water holding*" OR "material" OR "substrate*”). A total of 294 studies were found in Google Scholar (as of April 2019) through this literature search, of which 172 were excluded as irrelevant and a further 50 studies were excluded for being evaluations, grey literature or repeats of sources in the search. This resulted in a final 64 studies being suitable. This analysis identified 14 different habitats (Figure 5-1) with 15 different ecological enhancement habitat and study types identified (Figure 5-2). Several studies were counted twice where they conducted surveys over multiple habitat types. For the Web of Science search, n=7 results were found which were additionally contained within the Google Scholar search and so did not need additional counting (Evans et al., 2015; Firth et al., 2014a; Firth et al., 2014b; Hall et al., 2018; Loke et al., 2017; Naylor et al., 2017b; Ostalé-Valriberas et al., 2018). A full list of included studies can be found in Table A 5-1.

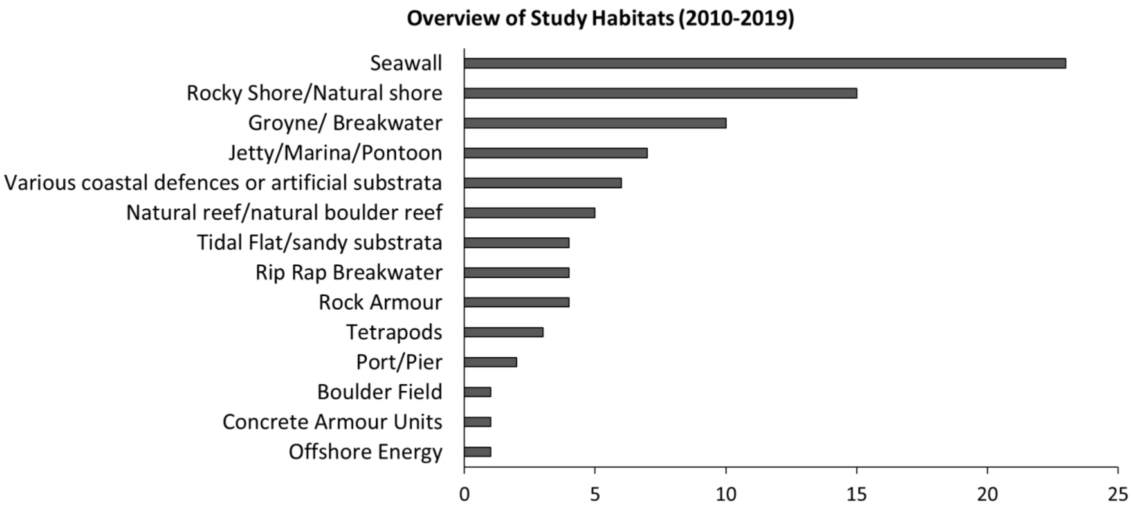


Figure 5-1. Overview of habitat types of studies from systematic review of ecological enhancement literature (2010-2019) (n=64 studies in n=86 habitat types).

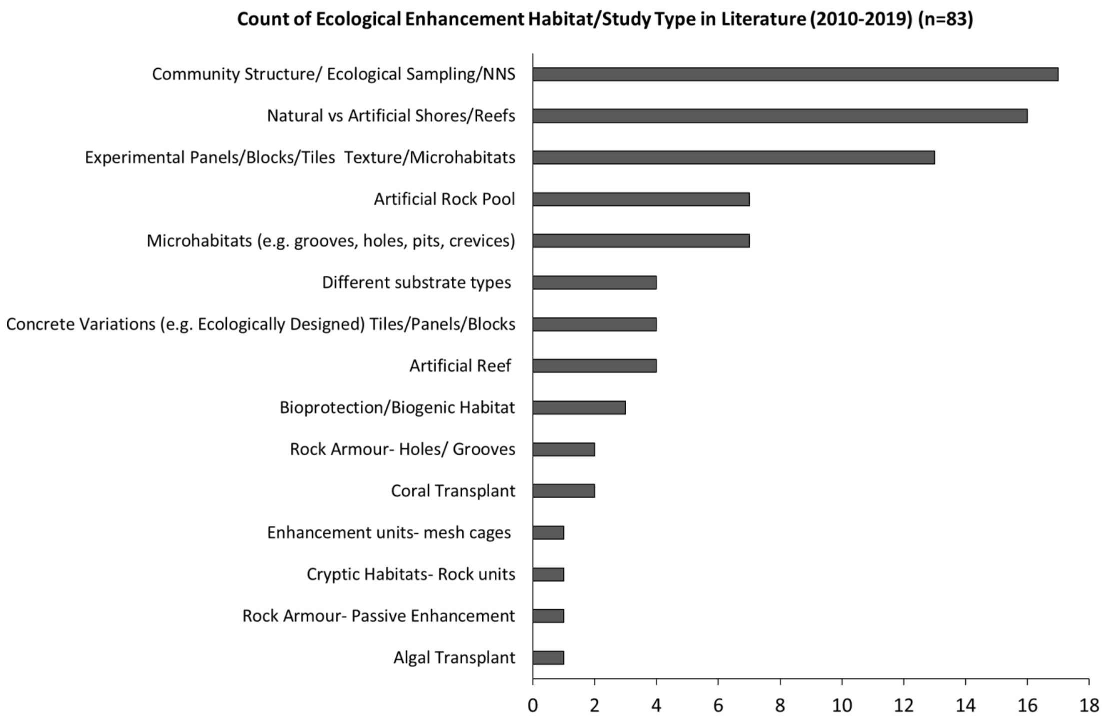


Figure 5-2. Ecological enhancement and habitat types identified in studies (2010-2019) (n=83).

Figure 5-1 and Figure 5-2 show a large body of research on actively designing or retrofitting artificial hard coastal structures to improve ecological outcomes (artificial rock pools, textured tiles, microhabitat creation and the like (Figure 5-2) (Coombes 2011; Evans et al., 2015; Firth et al., 2013). These demonstrate that ecological enhancements can deliver positive gains in ecosystem services (Strain et al., 2017b, not in review). Nevertheless, important gaps remain concerning the enhancement of rock armour revetments, as both passive enhancement strategies and retrofitting habitat is greatly overlooked (n=3 studies).

Boulder revetments are a key coastal defence (Bradbury et al., 1998; French, 2001), yet studies on the ecological value and suitability of rock armour (n=4) or rip rap breakwaters (n=4) are generally more limited (Figure 5-1) than other structures, such as seawalls (n=23). From this review, only Naylor et al. (2017) examined ecological enhancements at the design phase of construction (NB. Naylor et al. (2017) includes some of the research reported in this chapter). Few studies have compared lithology for ecological suitability (n=4) and so compared to studies examining active enhancements to improve the ecological suitability of coastal and marine engineering structures, limited research has examined the value of passive enhancements.

This chapter aims to address the above gaps in understanding through a combination of laboratory and fieldwork experiments. Firstly, laboratory tests were undertaken to determine the optimal combination of rock material properties to support early ecological colonisation. For this, albedo and water holding capacity (porosity and water absorption capacity) were used as proxies for temperature and desiccation stress and mm-scale roughness was calculated as this encourages early colonising species. Rock hardness was also tested since the affects both the engineering potential and ecological suitability. Following this, field experiments evaluated the effectiveness of low-cost passive enhancement strategies implemented at the onset of construction of a rock armour revetment in a high-energy wave environment at Hartlepool headland. At this site, an active enhancement trial was attempted but was destroyed by high-energy wave action. The high-energy wave context and the sites designation as a Ramsar site (JNCC, 2008), a site of special scientific interest (Natural England, 1997) and part of the Teesmouth and Cleveland Coast Natura 2000 site, meant that larger scale active enhancement was deemed unsuitable. As a result, passive enhancement options were explored to first identify an ecologically favourable rock material choice (out of the options available) and second to position individual revetment boulders with surface depressions facing upwards to replicate non-draining natural rocky shore features.

5.2.1 Research aims and hypotheses

The main aim of the field work was to determine if the passive ecological enhancement techniques of positioning and lithology resulted in differences in species richness and abundance when compared to randomly deployed, un-manipulated boulders (not expressly passively positioned, hereafter termed partially enhanced). Additional surveys were then carried out to compare the lighter Shap granite at the Hartlepool headland site with a nearby site using Larvikite (hereafter Norwegian granite) at Skinningrove of similar age (two years after installation). These sites were also examined for blast features (created during the quarrying process) to determine if these functioned as suitable habitats to increase the abundance and richness of species on boulders. Table 5-1 outlines the research questions and hypotheses: that both rock material properties and inherent rock surface complexity (rock mass properties) at a range of spatial scales (micrometres – decimetres) influences the early stage colonisation of coastal engineering infrastructure. These questions aim to establish the extent to which any passive enhancement of boulders in artificial coastal engineering structures is effective in facilitating early stage colonisation and so can produce tangible ecological benefits.

Table 5-1. Research questions and hypotheses.

Research Questions	Experimental Hypotheses
1. Which rock material properties are important for assessing the ecological suitability of boulders for coastal engineering?	**Lab experiments:** a) Lithologies that are softer with a combination of lighter colour, higher surface roughness and higher porosity will have greater ecological suitability. **Field experiments:** b) The lighter Shap granite performs better than the darker Norwegian granite for species richness and abundance. c) Carboniferous limestone has greater species richness and abundance than Shap granite.
2. Do natural surfaces outperform artificial?	a) Natural surfaces have greater species richness and abundance than artificial surfaces.
3. Which scale(s) of rock surface texture are most important for early stage (within 24 months) colonisation?	a) Mobile species abundance is greater on larger scale (dm-10s of dm's) features than smaller scale features (cm-dm). b) Early colonising barnacle abundance is unrelated to roughness in Shap granite or Norwegian granite.
4. Does passive positioning of features on rock armour improve ecological colonisation within 24 months?	a) Species richness and mobile species abundance is greater and algal abundance lower on enhanced boulders than on partially enhanced. b) Abundance of functionally important groups- barnacles and limpets on enhanced boulders peaks within 24 months.

	c)	Enhanced boulders match baseline biotope conditions within 24 months.
	d)	Abundance of key prey species (gastropods) for internationally important waterbirds is greater on enhanced boulders than on partially enhanced boulders.
5. Do quarried features (drill and blast holes) add ecological value and function similarly to active enhancement studies (pits and grooves)?	a)	Quarried features on boulders host greater species richness and abundance than adjacent surfaces.

5.3 Methods

5.3.1 Rock material choice- laboratory experiments

Laboratory tests were conducted by University of Glasgow undergraduate students (Breach, 2017; Harland, 2018; Scarr, 2018), with their individual data then collated and all subsequent analyses and results conducted personally. Ecological enhancement involves selecting ecologically suitable rock material for rock revetments and breakwaters in the design phase of construction. This includes the selection of light coloured, chemically favourable lithologies with rougher surfaces (µm-mm scale), which can maximise the ecological potential of artificial structures (Coombes et al., 2015; Kordas et al., 2014; Sempere-Valverde et al., 2018). Several lithologies commonly used in UK coastal protection schemes (Table 5-2) were compared for their surface roughness, surface albedo and effective porosity to identify the most ecologically suitable rock materials for future engineering designs. Tests were typically carried out on rough (broken) and smooth (cut) samples, with sample type specified in the results. These tests were later compared with existing published studies on the material properties of lithologies commonly used in coastal engineering to create an ecological suitability matrix.

Table 5-2. Lithology, location and characteristics for materials tests.

Lithology	Location Collected	Description
Shap granite	Hartlepool headland scheme, rock armour	Igneous rocks with pale grey, large pink crystals.
Norwegian granite	Skinningrove rock armour	Massive, dark grey igneous rock, with lighter grey crystals
Magnesian limestone	Hartlepool natural shore platform	Thickly bedded, yellowish beige dolomitic limestone,
Carboniferous limestone	Hartlepool headland scheme	Thickly bedded, medium grey limestone
Carboniferous limestone	Halkyn Quarry, North Wales	Thickly bedded, pale grey limestone
Blue Lias limestone	Glamorgan coast cliff, South Wales	Thinly bedded blue-grey Jurassic limestone

5.3.1.1 Surface Roughness

Micro-scale (µm-mm) surface roughness is important for successional communities to establish on natural rocky shores (Chabot and Bourget, 1988; Coombes et al., 2015; Green et al., 2012). Rock samples from each of the above six lithologies were scanned using a NextEngine laser scanner at 100µm resolution (Table 5-3). The only available weathered sample was of Magnesian limestone from the shore platform at Hartlepool. Scans were analysed in ArcMap and converted using an inverse distance weighted (IDW) technique to create a DEM surface. Surface roughness was quantified using standard deviation of slope. Pixel neighbourhoods were created around each DEM pixel to calculate standard deviation of elevations within this neighbourhood at the 25 mm^2 scale. This scale was optimal due to the small size of samples (maximum 6 cm^2).

Table 5-3. N numbers for surface roughness scans of each material type.

Lithology	Rough (broken)	Smooth (cut)	Weathered	Total N
Shap granite	9	3	0	12
Carboniferous limestone (Hartlepool)	3	3	0	6
Carboniferous limestone (Wales)	3	0	0	3
Magnesian limestone	3	3	3	9
Norwegian granite	6	0	0	6
Blue Lias limestone	3	0	0	3

5.3.1.2 Surface Albedo

Rock surfaces with low albedos result in increased thermal stress for colonising species as dark surfaces heat faster (McGreevy, 1985). Rock material samples from each of the six lithologies were placed on a control surface of white paper (albedo value of 0.65 (Gorski, 2011)). The top surface of each sample was photographed vertically with a Sony DSLR A390 to measure light levels. Total numbers of samples for each lithology varied with N=9 of each sample type (smooth and rough, dry) for Shap granite, n=6 for Norwegian granite and n=3 for Carboniferous limestone (Wales), Carboniferous limestone (Hartlepool), Magnesian limestone and Blue Lias limestone. A total of n=3 photos were taken of each smooth, rough and weathered sample in consistent lighting conditions and images loaded into ImageJ software. The mean value of brightness was determined from a pixel histogram and used in Equation 1, where Ks is mean sample brightness, Kp is mean paper brightness and CAp is the albedo paper value. Values were calculated for dry samples to avoid the low tide stress effect.

Equation 1: *Albedo (A) = (KsKp) x Cap*

5.3.1.3 Porosity

High porosity can reduce the engineering durability of materials by allowing the penetration of weathering processes (Coombes et al., 2011) so that lower porosities are favoured for engineering purposes. However, pore spaces reduce potential desiccation stresses and increase the ecological suitability for biota (Coombes et al., 2011; Green et al., 2012). Effective porosity was measured using equation 2 for pore volume and equation 3 for porosity calculations using n=10 samples of Shap granite, n=4 of Carboniferous limestone (Hartlepool) and Magnesian Limestones, n=3 of Blue Lias limestone and the Welsh Carboniferous limestone and n=6 of Norwegian granite.

Equation 2: *Pore Volume (Vv, cm^3) = (Msat-Ms)*

Equation 3: *Porosity (n, %) = ((Vv)/V) x 100*

5.3.1.4 Water Absorption Capacity

Water absorption capacity (WAC) was measured as the percentage uptake of water. Four 3 cm^3 unweathered blocks were cut from the centre of each lithology using a diamond tipped saw and a trim blade. Each cube was submerged in distilled water then cleaned and dried. Five of the cube faces were sealed using two coats of polyurethane yacht varnish, ensuring one-directional water movement on one face and mimicking deployed structures in the intertidal zone. Sealed cubes were then oven-dried for 24 hours at 105°C then placed into a desiccator and allowed to cool to room temperature (~21°C) for 2 hours. Volume (V) and dry mass (Ms) were calculated for each cube before submerged in seawater at room temperature for one week before removal and reweighing. WAC was calculated using equation 4.

Equation 4: WAC (%) = (Msat-Ms) x 100

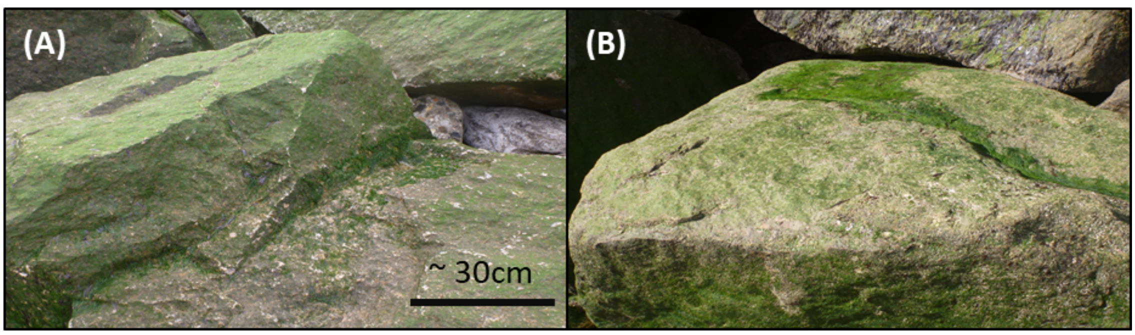
5.3.1.5 Rock Hardness

The small physical size of the rock samples precluded the use of a Schmidt hammer hardness test and so rock hardness data was gathered following the method of Hoek and Brown (1997). This method allowed for an insight into rock hardness, with rock samples of similar small sizes tested by scratching with a pen knife and/or repeated blows of a geological hammer to determine a field estimate of strength and an associated grade from extremely weak to extremely strong. Additional samples of Cornish granite and Portland limestone used in Coombes (2011) were tested in this experiment to include them in summary findings and engineering recommendations as other rock material properties had already been tested.

5.3.2 Study Sites

Two study sties were selected: Hartlepool (54°41'48.3"N 1°10'31.4"W) and at Skinningrove (54°34'22.7"N 0°54'00.2"W) 14.4 miles southeast of Hartlepool. Hartlepool headland is a relatively exposed, high-energy site protected by 150 year old defences that are being replaced due to poor condition and frequent overtopping. Construction of the Hartlepool headland coastal defences commenced in 2015 to protect 562 residential and commercial properties. The defences include an enhanced textured seawall using Reckli formliners to both improve the structural complexity of the wall and mimic natural rock providing up to 27 mm deep textured relief (Naylor et al., 2017b). In front of the seawall, an 800x10 m rock armour revetment of 8-13 tonne Shap granite boulders was finished in late 2017. Some boulders were passively positioned to favour features such as ledges that generally ranged from 14.05-118.66 cm (average height of 40.77 cm ± 6.74 SE, n=14 boulders) (Figure 5-3) or depressions that could retain water and classified here as enhanced boulders. Other boulders that were randomly deployed were classified as partially enhanced (Figure 5-3) due to the base level of enhancement provided by choice of the enhanced rock material. The defences are underlain and fronted by a Magnesian limestone intertidal shore platform. The scheme incorporated passive enhancement to limit the impact of construction on the prey species (e.g. *Patella vulgata*) for key waterbirds at the site (Naylor et al., 2017b). The comparator site at Skinningrove (Figure 5-4) is a moderately exposed site that was previously at risk of flooding due to the deterioration of existing defences. The construction of a 310x10 m Norwegian granite rock armour revetment in 2015 provided a comparable installation of similar age close to the Hartlepool scheme.

Figure 5-3. (A) Example of ledge on enhanced boulder and (B) partially enhanced boulder, lacking features. Full boulders approximately 235.18 cm ± 10.22 S.E. wide at Hartlepool (n=31).



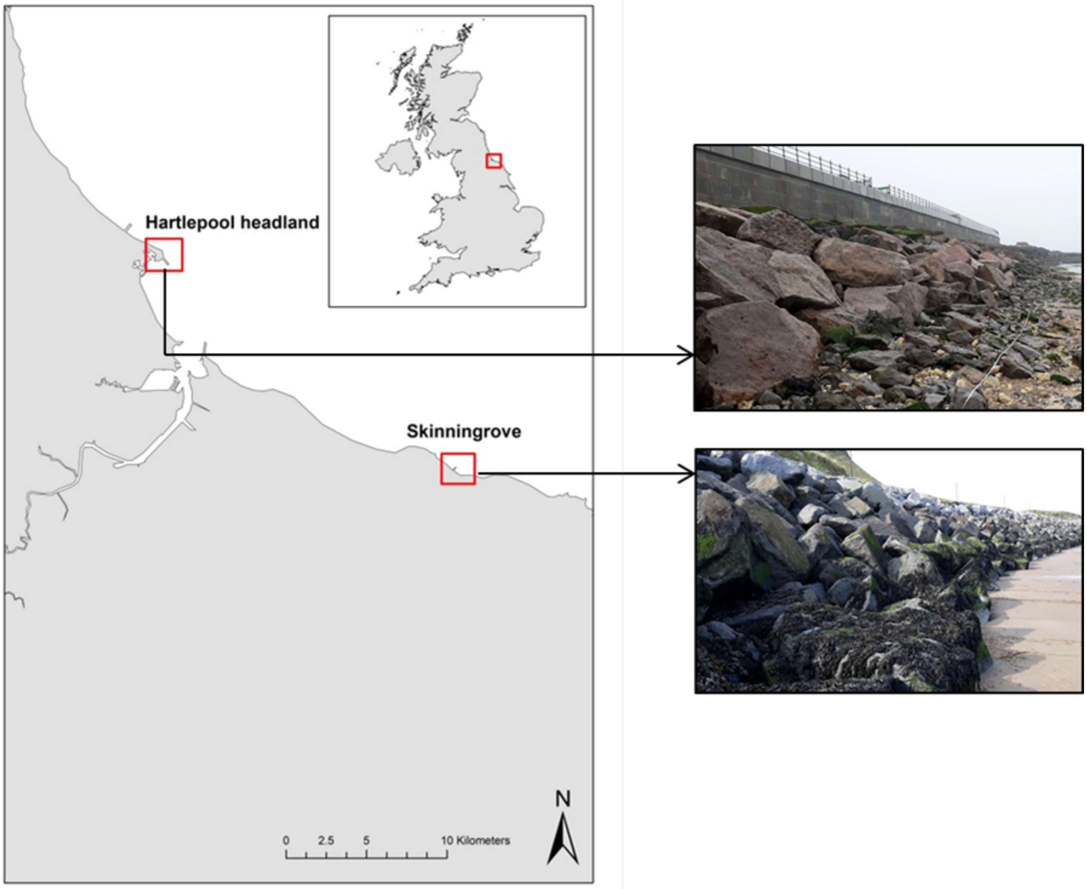


Figure 5-4. Locations of Hartlepool and Skinningrove. At Hartlepool, Shap granite was used with passive enhancement. Skinningrove used Norwegian granite without any passive enhancement. Scale of individual boulders approximately 235.18 cm ± 10.22 S.E. wide at Hartlepool (n=31) and 143.59 cm ± 12.75 SE at Skinningrove (n=10).

At Hartlepool, surveys were conducted along the north east facing section of the intertidal rock armour revetment. Staged installation produced a data time series at Year 0, Year 1 and Year 2 years after installation (hereafter YAI) (Table 5-4). During monitoring, more sections were added to the rock armour revetment which meant that in 2017 there was a time series from Year 0 to Year 2 (Figure 5-5). Field trials to establish baseline ecological conditions prior to construction were repeated to identify the post installation ecological responses of intertidal species to passive enhancement treatments at Hartlepool at Year 1 and Year 2. The Skinningrove surveys aimed to evaluate how Shap granite, selected at Hartlepool due to its coarse-grained and fine-scale (mm-cm) and large-scale (dm – m) roughness and light colour, compared to the smoother and darker Norwegian granite of similar age at Skinningrove.

Table 5-4. Location, date and study type for all surveys conducted at Hartlepool and Skinningrove.

Site	Date	Tidal Level	Years After Installation (YAI)	Study
Hartlepool	September 2016	Upper	-	Baseline habitats (natural and artificial)
	October 2016	Upper	Year 0 Year 1	Enhanced quadrat (EQ) vs Partially enhanced quadrat (PEQ) Granite and limestone
	October 2016	Mid-Upper	Year 0 Year 1	EQ vs PEQ
	June 2017	Mid-Upper	Year 0 Year 1 Year 2	Enhanced (E) vs Partially Enhanced (PE) boulders
	September 2017	Upper	Year 2	E vs PE boulders Shap granite and Carboniferous limestone
	September 2017	Mid-Upper	Year 2	Hartlepool Shap granite vs Skinningrove Norwegian granite
Hartlepool and Skinningrove	September 2017	Mid-Upper	Year 2	Boulders with quarried features and adjacent boulders

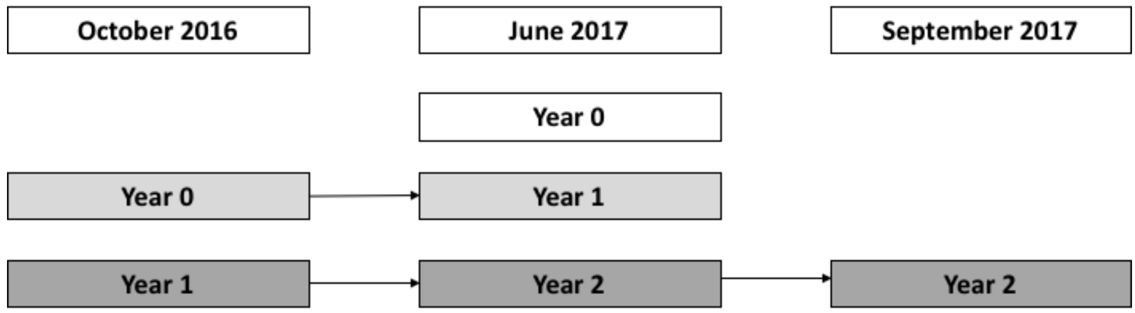


Figure 5-5. Schematic of how the “years after installation (Year 0, Year 1, Year 2)” designation changed over time: rock armour was installed in sections, so each sampling would show some sections had colonised and some were new structures. So those that start at Year 0 become Year 1 later with a new section of Year 0 added in a different part of the site.

5.3.3 Experimental Design

5.3.3.1 Baseline monitoring

At Hartlepool, several natural and artificial habitats were sampled along the length of the scheme. However, access and funding constraints at the pre-build phase followed by a rapid construction timetable, prevented ecological baseline surveys of the natural platform and artificial surfaces prior to construction in year 1. Subsequent surveys of the Magnesian limestone natural shore platform fronting the rock armour at Hartlepool were conducted on exposed shore platform surfaces that were not yet covered or damaged by machinery. These surveys on natural surfaces, as well as surveys on artificial surfaces, were conducted in September 2016, with all sampling occurring in the upper tidal levels (0-10 m from the seawall) due to safety restrictions.

In September 2016, surveys using 25x25 cm quadrats were conducted in six different natural and artificial habitats, including the concrete wall and natural horizontal shore platform, to provide baseline data for the site. To ensure that surveys were representative of the entire area, some habitats were sampled across multiple points along the revetment, referred to as plots. The natural shore was surveyed at four plots along the extent of the existing revetment (Year 0 and Year 1 as of September 2016) (Figure 5-6). Figure 5-6 shows the range of habitat types examined and divides surveys into artificial (concrete seawall and rock armour) and natural (shore platform).

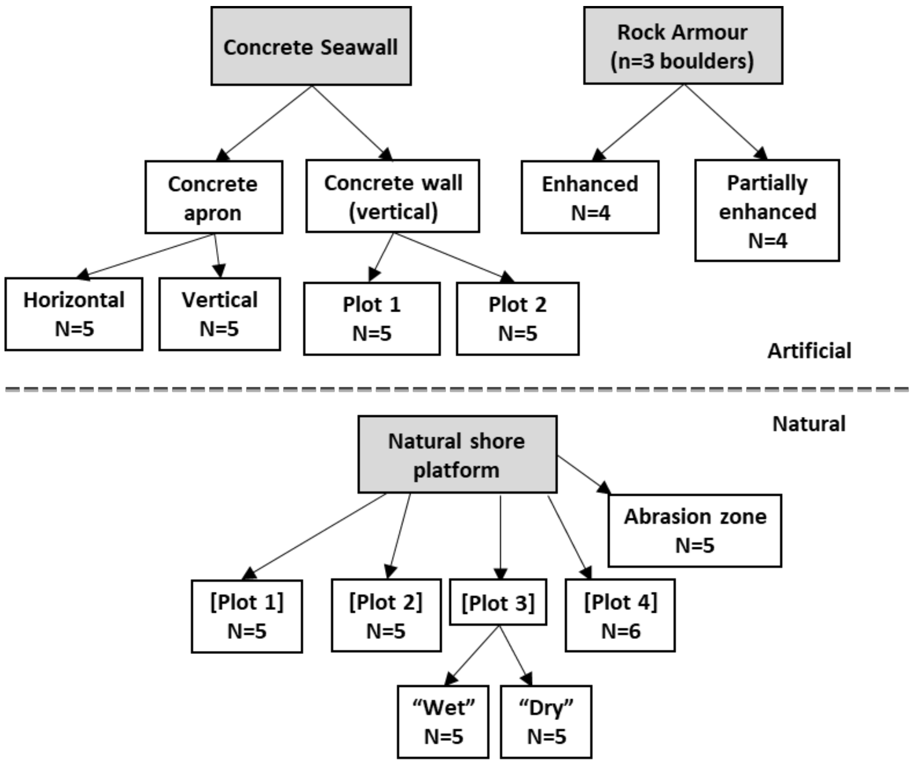


Figure 5-6. Schematic of the baseline survey design (September 2016) in both artificial and natural habitats, with N indicating number of quadrats.

Quadrat spacing in each plot was at least 50 cm (Chapman and Bulleri, 2003; Firth et al., 2013; Moreira et al., 2006), with counts for mobile species and visual estimates of percentage cover for attached species (algae) and barnacles collected in each quadrat. The abbreviation "sp(p)." refers to either one or several unidentified species. Analysis was undertaken at the species level where possible, otherwise identification was done at a higher taxonomic level.

Areas of shore platform habitat were relatively restricted (e.g. <20 m long) resulting in each plot being between 5 and 8 m long and spaced 10 m apart (after Chapman and Bulleri, 2003; Moreira et al., 2006). Results from the September 2016 surveys of the natural horizontal platform (n=26 quadrats) were used as the control baseline to compare the rock revetment against subsequent sampling. In addition, data from the MarClim project (Marine Biological Association, 2019) from a 2008 survey of the Hartlepool headland was used to identify the range of species present prior to construction disturbance.

5.3.3.2 Rock armour sampling- October 2016 (Year 0, Year 1)

Three sites, 30-40 m apart, were sampled in October 2016 along the existing rock armour (Chapman, 2012; Green et al., 2012). Sampling was primarily on Shap granite boulders, with Year 1 boulders deployed in 2015 and Year 0 boulders deployed in 2016. Carboniferous limestone boulders were also sampled along the upper transect where they were present (n=3 Year 1). There were too few limestone boulders at the upper tidal level for a robust sample, so the survey simply offers a comparison of material types. A 20 m transect consisted of tape measures laid horizontally along the top and bottom rows of the rock armour to guide sample selection. The upper transect was situated just below the splash zone and the lower level transect was placed in the mid-upper zone, with boulders selected from the bottom two rows of the rock armour installation. Although previous studies have taken samples across 4 m wide transects (Ríos and Mutschke, 1999), the size of the boulders meant this was not realistic. Sampled boulders were at least 1 m apart (after Green et al., 2012).

A random number generator was used to select five sampling points along each 20 m transect and enhanced and partially enhanced boulders were selected within a 1 m radius of each sampling point (see Table 5-5). Year 0 had a lower number of boulders surveyed as this section was still under construction, restricting the sampling area. Two quadrats (25x25 cm) were placed on enhanced and partially enhanced sections of enhanced boulders, with quadrats spaced at least 50 cm apart, and one 25x25 cm quadrat placed on partially enhanced boulders. For Year 1, a total of n=10 of each enhancement type was surveyed at each height (after Green et al., 2012) as boulders typically measured between 2-3 m across and so this sampling size offered a representative sample at multiple points and intertidal heights along the rock armour installation. Attempts were made to mark the corners of quadrats for succession studies with marine epoxy and bingo chips, but these were destroyed by wave activity.

Table 5-5. Sampling 'N' numbers for October 2016 at Hartlepool.

Mid-Upper Tidal Level		
	Year 0	**Year 1**
Enhanced granite	5	10
Partially enhanced granite	5	10
Carboniferous limestone	0	0
Upper Tidal Level		
Enhanced granite	5	10
Partially enhanced granite	5	10
Carboniferous limestone	0	3

5.3.3.3 Development of method for rock armour surveying (post-October 2016)

Traditional ecological sampling using quadrats is a proven method of survey (Green et al., 2012; Le Hir and Hily, 2005; Sousa, 1979). However, the size of boulders at Hartlepool mostly exceed two metres (width/ b-axis) and quadrat sampling was not representative of species present and feature types. As a result, a new method was developed for subsequent surveys to span physical scales (cm through dm's) and better quantify the abundance and richness of species around key habitat sub-features on the rock armour. This in turn would better identify links between geology, geomorphology and biodiversity. This method was refined with reference to the methods in

Table 5-6. The new method was conducted by visually sampling the entire top surface of boulders since a 25x25 cm area would risk misrepresenting the diversity and importance of habitat features, such as ledges, which did not fit into quadrats. Counts of individuals of mobile species and percentage cover of attached species and barnacles were recorded. The presence and location of mobile species was also recorded on boulder surfaces and on/in specific features. The count and percentage cover of each habitat feature type (crack, crevice, pool, ledge, other) was also recorded.

Partially enhanced boulders were formally defined as boulders with less than 20% of their surface 'enhanced' by positioned features, such as ledges, pools and cracks. This was taken as a threshold value for all further field experiments after sampling of boulders in June 2017 when visual estimations of the average surface area enhanced (%) on both boulder types was recorded. Enhanced boulders had an average of 37.19% (±2.36 SE) enhanced surface (n=40 boulders) whereas partially enhanced boulders had an average of 6.00% (±0.78 SE) enhanced surface (n=40 boulders).

Table 5-6. Sampling strategies of ecological studies examining intertidal boulder fields and other artificial coastal structures.

Location	Sampling Area	Time Data	Method	Sampling Size (replicates)	Reference
Southern California	Rocky intertidal boulder fields	0-2 years (up to 24 months post-clearing)	Random sampling along horizontal transects at two tidal levels	20 per tidal level	Sousa (1979)
South east Australia	Two boulder fields	October to November 1982	Randomly selected boulders at high and low shore levels	8 per size class	McGuinness (1984)
New South Wales, Australia	Intertidal and subtidal boulders	0,4,8 and 12 months	Position of marked boulders in 80x20 m field mapped using tape measure around site	20 per treatment	Chapman and Underwood (1996)
Western Brittany, France	Intertidal boulder fields	April 1995	Two level stratified sampling, 3 different strata Sampled microstrata, e.g. open rock surfaces and sheltered overhangs, summed their surfaces to estimate complexity	3 per strata	Le Hir and Hily (2005)
Southwestern Japan	Intertidal boulder field	28 day intervals over 11 months 2002-2003	Field was 200 m wide and 50 m seaward Minimum distance 1.2 m between stones	60 sorted into size classes	Londoño-Cruz and Tokeshi (2007)
Plymouth Breakwater, UK	Homogeneous intertidal rock pools on breakwater	13 months	36 pools in total. Two pools on upper surfaces of concrete blocks, separated by 1.5 m	4 per treatment	Griffin et al. (2010)
Sydney Harbour, Australia	Lower intertidal boulder field	0-10 months	Two 100x100 m basalt fields and two sandstone fields sized 200x20 m and 100x20 m were divided into three sites and separated by 10 m, with boulders at least 1 m apart	10 per site	Green et al. (2012)

5.3.3.4 June 2017 sampling (Year 0, Year 1, Year 2)– Ecological impact of passive enhancement of features

This study aims to establish whether any differences in species abundance and richness occurred between enhanced and partially enhanced granite boulders at Hartlepool. The rock armour installation was split into three sampling areas depending on YAI (Year 0, Year 1, Year 2) (see Figure 5-5 for a schematic view). A minimum of 20 m separation between sampling areas ensured representative sampling of the rock armour installation (after Green et al., 2012). Stratified random sampling was conducted along 60 m horizontal, shore-parallel transects laid out along the lowest row of the rock armour at each site and 15 points selected by random number generators along each transect. The closest partially enhanced/enhanced boulders to each point were then sampled, with a spacing of at least 1.5 m apart (Griffin et al., 2010; Londoño-Cruz and Tokeshi, 2007).

Fifteen boulders of each enhancement type (n=30 total per transect) were selected and sampled in Year 0 and Year 1 areas, and, due to time and safely constraints due to tidal conditions, only ten boulders per enhancement type were retrieved in Year 2 (n=20 total). This was deemed an appropriate number of replicates based on other studies (

Table 5-6). Only boulders with top surface angles of less than approximately 45^{o} were sampled. Surfaces steeper than this were deemed to not have been properly enhanced as they would not retain water due to runoff.

5.3.3.5 September 2017 (Year 2)

5.3.3.5.1 Hartlepool – upper tidal level material and enhancement comparison

A 60 m horizontal transect was laid out across the uppermost row of boulders in the Year 2 section of the installation. Fifteen boulders of each enhancement type were surveyed as well as an additional n=15 Carboniferous limestone boulders for a comparison of rock materials at the upper tidal level. The sampling strategy used here was replicated from the June 2017 sampling methods outlined above.

5.3.3.5.2 Hartlepool and Skinningrove- rock material field comparison study

This study is a field test of two of the rock materials used in laboratory tests to determine whether lithology influences intertidal ecology at the two sites of similar age. This experiment compared ecological surveys of the Year 2 Hartlepool Shap granite and the similarly aged Skinningrove Norwegian granite (2 YAI). Spacing between boulders and stratified random sampling methods were consistent with the June 2017 experimental design.

5.3.3.5.3 Habitat value of quarried features

Blastlines and blastholes (Figure 5-7), often found in quarried boulders, occur at a scale typical of retrofitted active ecological enhancements (cm-scale depth). Research question 5 (Section 5.2.1.) aimed to establish whether these quarried features added ecological value and function. Due to the scattered occurrence of these features, sampling occurred along the full length of the Year 2 section of the revetment at Hartlepool and Skinningrove. Boulders displaying these features were sampled, as well as any unadorned boulder immediately to the left. This resulted in sampling of n=6 blastline and blasthole adorned boulders at Hartlepool (Year 2), n=18 blastline and blasthole adorned boulders at Skinningrove (Year 2), together with an equal number of unadorned "boulders to the left".

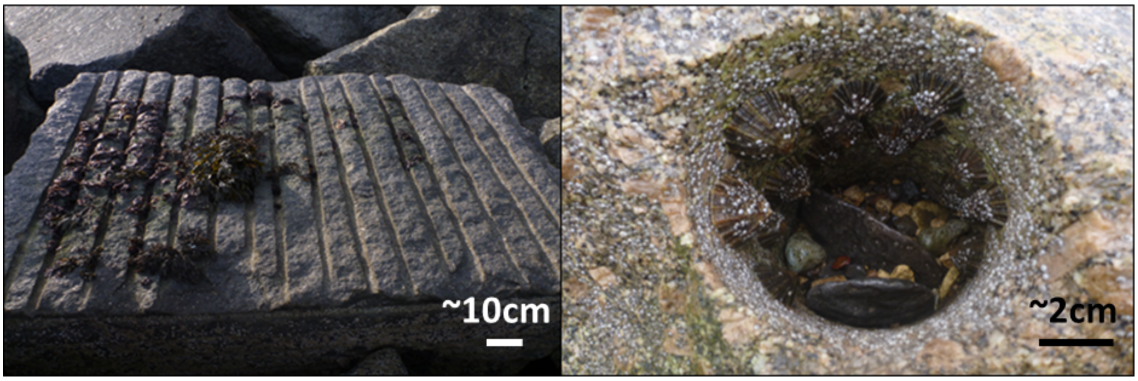


Figure 5-7. (A) Blastlines at Skinningrove (B) Blasthole at Hartlepool with approximate scale added in cm.

In addition to this, the rock surface of one boulder with blastlines at Skinningrove and one boulder with ledge habitat at Hartlepool was scanned using a TLS Leica C10 with a vertical and horizontal resolution of 3 mm at 2 m distance. These scans were processed in Leica Cyclone 9.0 and the point clouds cleaned, merged and exported to ArcMap to generate a DEM. From this, profile graphs of the boulder surfaces were created using the 3D Analyst toolbar to highlight the differences in scale of these features.

5.3.4 Statistical Analyses

Data were tested for normality using Shapiro Wilk's test and for the majority of tests run, the distribution of species richness and abundance data was significantly non-normal ($p<0.01$). Therefore, non-parametric Kruskal-Wallis ANOVAs were used throughout this chapter to examine differences between the ecology on enhanced and partially enhanced boulders, between YAI, between enhancement types within YAI and between rock material types. Kruskal-Wallis was also selected due to the variation in sampling sizes within each study. Multiple Kruskal-Wallis tests were conducted on species richness and mobile, algae and barnacle abundance data. Where significant differences were found, Dunn's post hoc tests for pairwise comparisons were conducted on each pair of groups, with Bonferroni adjusted p-values for multiple comparisons to control the familywise error rate. Analyses were carried out in R version 3.5.1 (R Development Core Team 2018).

5.4 Results

5.4.1 Rock material tests- Laboratory study

Porosity significantly differed between lithologies ($H[5]=13.688$, $p=0.018$) but post-hoc testing was not significant with the adjusted p-values for multiple comparisons ($0.05/10=0.005$). However, it is evident that the Magnesian limestone has a higher average porosity than all other lithologies examined, particularly compared to the two granites deployed at the field sites surveyed (Figure 5-8). Similar patterns were observed with water absorption capacity (WAC), which differed significantly between lithologies ($H[5]=13.942$, p= $p=0.0160$) but not in post-hoc testing following adjustments for multiple comparisons. Magnesian limestone had the greatest average WAC (8.907 % ± 0.152 SE) compared to other lithologies, with the lowest WAC being tested in the Shap (0.188% ±0.079 SE) and Norwegian (0.171% ± 0.110 SE) granites.

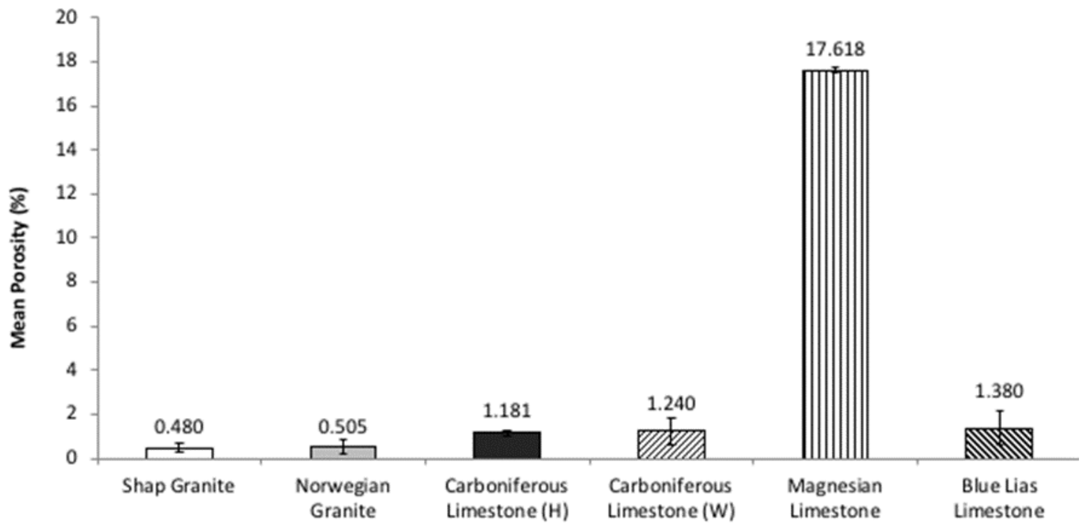


Figure 5-8. Mean porosities of the six lithologies, taken from n=10 samples of Shap granite, n=4 samples of Carboniferous (Hartlepool) and Magnesian limestones, n=3 samples of Blue Lias limestone and Carboniferous (Welsh) limestone and n=6 samples of Norwegian granite ($\bar{x}$ ± standard error).

Dry surface albedo was examined for grouped smooth, rough and weathered rock samples as there was no significant difference between the rough, smooth and weathered albedo ($H(2)=2.659$, $p=0.265$). Differences in albedo were observed between the six lithologies examined ($H[5]=49.192$, $p<0.001$). Magnesian limestone was lighter in colour and more reflective than the Blue Lias limestone ($z=-4.077$, $p<0.001$) and Norwegian granite ($z=6.069$, $p<0.001$) samples. Norwegian granite was the darkest lithology sampled with the Shap granite and two Carboniferous limestones (Welsh and Hartlepool) significantly more reflective ($p<0.01$) (Figure 5-9). Only the Blue Lias limestone did not significantly differ from Norwegian granite, this lowers the ecological suitability of both lithologies in a warming climate.

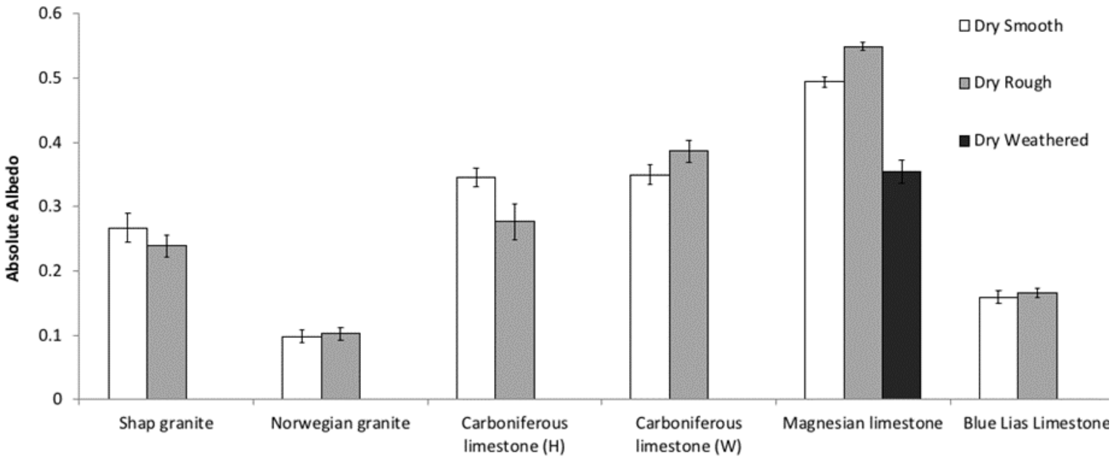


Figure 5-9. Surface albedo for dry smooth (cut), rough (broken) and weathered (Magnesian limestone only) samples of each of the n=6 lithologies. These were from n=9 Shap granite, n=6 Norwegian granite samples (smooth/rough) and n=3 for the remaining lithologies ($\bar{x}$ ± standard error).

Different methods of preparing the samples were analysed separately as there was a difference between rough, weathered and smooth samples ($H(2)= 20.338$, $p<0.001$), with both rough ($z=4.490$, $p<0.001$) and weathered ($z=2.324$, $p<0.05$) samples rougher than smooth samples. There were no significant differences in surface roughness when comparing all rough (all lithologies, $H(5)=10.703$, $p=0.058$) or smooth samples (only for Shap granite, Hartlepool Carboniferous limestone or Magnesian limestone, $H(2)=1.867$, $p=0.393$) between lithologies. To make recommendations on ecological suitability of rock materials for boulders, the limestones and the granites were subsequently analysed separately.

There was little difference in the roughness of rough samples of the Shap and Norwegian granite at the 25 mm^2 scale ($H[1]=1.005$, $p=0.316$). In comparing the different limestones, there was a significant difference in the surface roughness between lithologies ($H(3)= 8.077$, $p=0.044$) and although post-hoc testing was not significant following adjusted p-values (0.05/6=0.008), there was a notable difference between the Welsh Carboniferous limestone and the other examined limestones (Figure 5-10), making the Welsh Carboniferous limestone likely to be more ecologically suitable at the 25 mm^2 scale.

In individually examining the Magnesian limestone, which included weathered samples (Figure 5-10)., the weathered rock was rougher than the smooth sample at the 25 mm^2 scale examined (z=2.683, p=0.011). No significant difference was detected between the rough samples (broken rock) and the weathered samples but at the 25 mm^2 scale, the mean standard deviation of slope increases by 103.93% between rough and weathered samples. This showcases the potential of Magnesian limestone to become more ecologically suitable over time as the rock gets rougher with exposure to the marine environment (Figure 5-10).

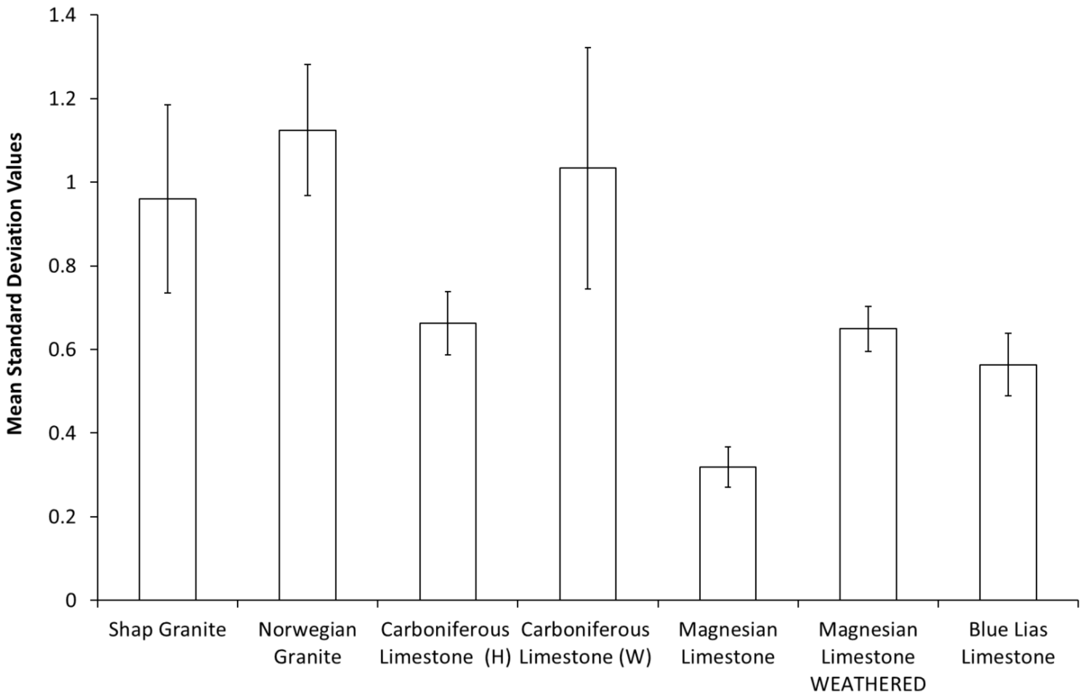


Figure 5-10. Mean standard deviations of slope (surface roughness) values for rough samples of examined lithologies and weathered Magnesian limestone ($\overline{x}$ ± standard error).

Rock hardness is listed in Table 5-7 and shows the granites to be the hardest lithologies and that within the limestones, the Carboniferous (Welsh) limestone was the most suitable for engineering purposes as it did not crumble or fracture with ease.

Table 5-7. Rock hardness grades as gathered by repeating experimental methods of Hoek and Brown (1997).

Lithology	Hardness Grade	Meaning
Shap granite	R5	Fractured after repeated blows with a geological hammer
Norwegian granite	R6	Only chipped with a geological hammer
Cornish granite	R5	Fractured after repeated blows with a geological hammer
Carboniferous limestone (Hartlepool)	R1	Crumbled after being hit with the point of a geological hammer
Carboniferous limestone (Welsh)	R4	More than one blow of a geological hammer is required to fracture the rock
Magnesian limestone	R2	Firm blow with point of geological hammer left shallow indentation
Blue Lias limestone	R1	Crumbled after being hit with the point of a geological hammer
Portland limestone	R1	Crumbled after being hit with the point of a geological hammer

5.4.2 Baseline (September 2016)

The pre-construction baseline survey conducted by MarClim in 2008 (Marine Biological Association, 2019) recorded a total of 13 species at the headland site (Table 5-8). Of these species, algae contributed the most to species richness (7 species), followed by gastropods (3 species) as listed in Table 5-8. This is similar to nearby rocky shores sites at Seaham (11 species) and Roker (10 species) (Marine Biological Association, 2019).

Surveys in September 2016 recorded 18 species across the shore platform, an average of 10 species per sampling plot. Abundance results mirrored the earlier MarClim survey with *Patella vulgata* recorded in high densities on the sampled shore platforms (average of 33.23/m^2). Several species absent in 2008 were common in 2016 (based on SACFOR ranking in Burrows et al., 2008), including *Littorina saxatilis* (n=5.76/m^2) and *Anurida* *maritima* (n=12/ m^2). From comparing the early MarClim survey data with data collected during 2016 surveys, more species were recorded in 2016 but both newly recorded species and those that were present in the original surveys were found to be in lower abundances (Table 5-8).

Table 5-8. MarClim survey results at Hartlepool (2008) (✓) with additional species recorded in 2016 baseline surveys (❖)

Species	Super Abundant	Abundant	Common	Frequent	Occasional	Rare
Laminaria digitata					✓	
Fucus spiralis		✓				
Fucus vesiculosus	✓		❖			
***Ulva* (spp.)**			❖			
Fucus serratus		✓				
Mastocarpus stellatus				✓		
Chondrus crispus				✓		❖
Palmaria palmata			✓			
Actinia equina		✓				❖
Semibalanus balanoides		✓			❖	
Mytilus edulis						✓❖
Patella vulgata	✓		❖			
Littorina littorea				✓		❖
Littorina obtusata					❖	
Littorina saxatilis					❖	
Melarhaphe neritoides						❖
Nucella lapillus					✓	
Polydora ciliata						❖
Talitrus saltator						❖
Rhodothamniella floridula					❖	
***Lithothamnion* (sp.)**						❖
***Verrucaria* (sp.)**					❖	
Anurida maritima						❖
Osmundea pinnatifida						❖

Sampling was undertaken on the vertical concrete apron (CAV), vertical concrete seawall (CWV), horizontal concrete apron (CAH), horizontal natural platform (PH), abraded horizontal platform (PHA) and rock armour (RA) (enhanced and partially enhanced boulders) habitats. This resulted in a total of n=21 comparisons so p-values were adjusted accordingly (0.05/21=0.0024). Species richness was found to be significantly different between habitat types (*H*[6]= 34.245, p<0.001), with CAV having a greater species richness than the abraded horizontal platform (z=3.709, p=0.0022). Both mobile abundance (*H*[6]= 19.575, p=0.003) and algae and lichen abundance (*H*[6]= 15.487, p=0.017) showed a difference between habitat types but on further post-hoc testing these were not significant due to the high number of pairwise comparisons made.

5.4.3 Baseline- shore platform and rock armour comparisons

From here on, comparisons are made for species richness and abundance between the shore platform and the rock armour replacements. Results are split into natural horizontal shore platform surveys (SP1,2,4), wet and dry shore platform (SP3 wet, SP3 dry) and rock armour (enhanced (E) and partially enhanced (PE) boulders). Species richness (*H*[4]= 14.244, p=0.007) and algae and lichen abundance (*H*[4]= 11.536, p=0.021) differed between habitat types but post-hoc comparisons were not significant under adjusted p-values (0.05/10=0.005). Mobile abundance also differed between habitat types (*H*[4]= 19.521, p= 0.001), with the shore platform surveys having greater mobile species numbers than dry platform quadrats (z=3.514, p=0.002). In Figure 5-11, enhanced boulders (6.5± 0.29 SE) performs similarly to the PH in terms of mean species richness (6.75 ± 0.48 SE), with both exceeding partially enhanced boulders (4.5±0.29 SE). The wet shore platform performs equally well as the enhanced boulders (6 ± 0 SE), but dry habitat underperforms (4.4 ± 0.25 SE) and is more in line with partially enhanced boulders. Similar patterns can be seen with mobile abundance, with the horizontal platform performing in line with the enhanced boulders and the dry platform having lower mobile species abundance than all other habitats surveyed. Algae and lichen abundance was highest on the dry platform and partially enhanced boulders, this could suggest that other habitats are more successionally developed.

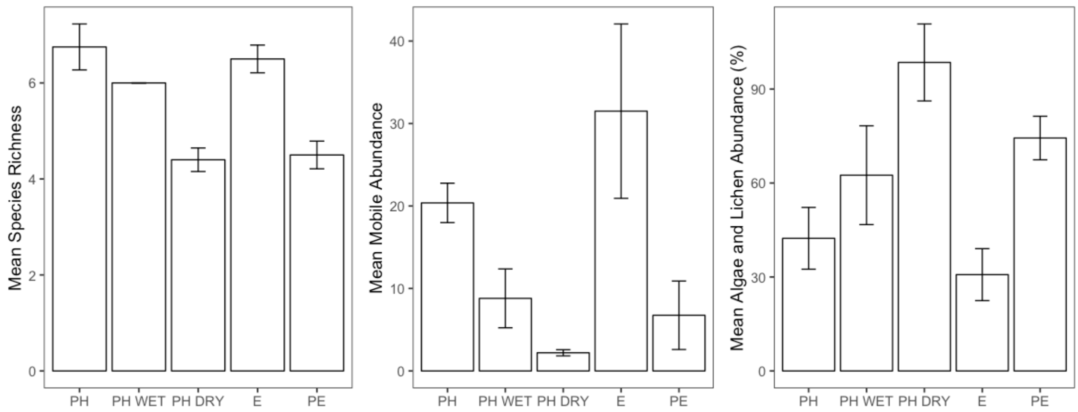


Figure 5-11. Mean species richness, mobile and algae/lichen abundance in horizontal natural platforms (PH) (n=16 quadrats), platform wet (n=5), platform dry (n=5) and enhanced (n=4) and partially enhanced (n=4) quadrats on boulders ($\bar{x}$ ± standard error).

Nineteen taxa were recorded across the shore platform and rock armour habitats surveyed, with nine species unique to the natural shore platforms and *Porphyra umbilicalis* unique to the rock revetment. In examining the number of species per plot (individual shore platforms included), lower species richness can be seen on the rock revetment (average of 8 species combined) than shore platforms (average of 10 species combined) (Figure 5-12).

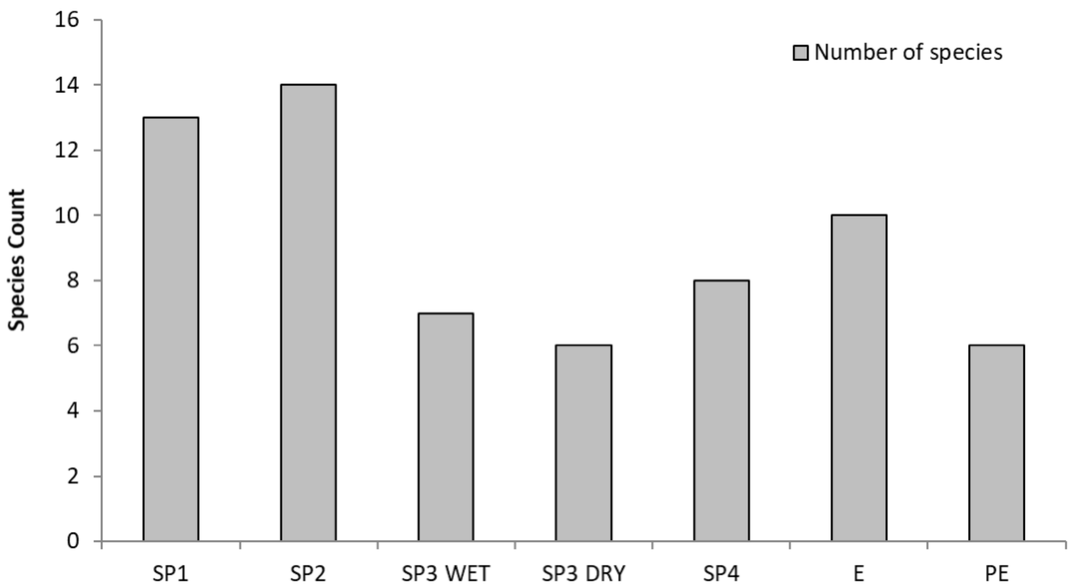


Figure 5-12. Species richness recorded on each shore platform plot (SP1-4), enhanced (E) and partially enhanced (PE) boulders. Breakdown of shore platform numbers can be seen in Figure 5-6.

As *Patella vulgata* is an important bird prey species for this site and the one of the targeted species for enhancement, it was individually examined and found that there was a difference between habitat types (*H*[4]= 20.909, p=0.0003). No difference was detected between enhanced boulders and the horizontal platform. In contrast, partially enhanced boulders had fewer limpets than the horizontal platform (z=-3.485, p=0.002). Other comparisons were not significant with adjusted p-values (0.05/10=0.005). On the shore platforms surveyed, an average of eight limpets occurred compared to an average of six individuals on combined rock revetment data. In contrast, when splitting the revetment data between enhanced and partially enhanced boulders (Figure 5-13), *P. vulgata* averaged 12 individuals on enhanced boulders compared to one on partially enhanced boulders, which is more in line with the natural rocky shore baseline.

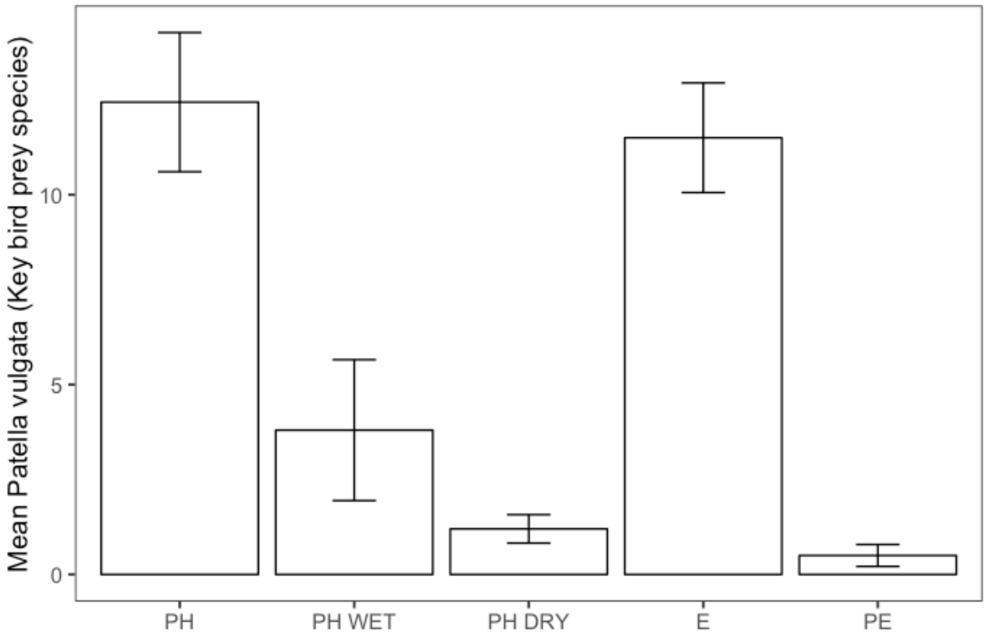


Figure 5-13. Mean limpet abundance on shore platform, PH=platform horizontal and rock revetment habitats, E= enhanced and PE=partially enhanced boulders (September 2016) ($\bar{x}$ ± standard error).

After *P.vulgata, A. maritima* and *L. saxatilis* had the next highest abundances, with the baseline shore platform having greater abundances than newly colonising rock revetment. The abundance of species varied greatly by plot, with *A. maritima* occurring in the greatest numbers in SP1 (average of 5 individuals ± 2.24 SE), SP2 (10± 3.54 SE) and enhanced boulders (7.5 ± 7.5 SE). Similarly, *L. saxatilis* varied by plot, with this species recorded in SP1-3 and in n=2 quadrats on enhanced boulders, with the greatest averages observed in SP1 (n=1.6 ± 1.36 SE), wet quadrats in SP3 (4.4 ± 1.4 SE) and enhanced boulders (3 ± 2.68 SE) which greatly contrasts the 0.8 (± 0.37 SE) individuals observed on average in dry plots in SP3. This highlights the important of water-retaining features as a determinant of species abundance.

For *S. balanoides*, abundance was significantly different between habitat types (H[4]=30.393, p<0.001). Post-hoc Dunn's tests with adjusted p-values (0.05/10=0.005) revealed that enhanced and partially enhanced boulders had greater abundance of barnacles than the dry platform (z=4.131, p=0.0002 and z=3.590, p=0.002 respectively). Both enhanced and partially enhanced boulders also had greater barnacle numbers than the wet shore platform (z=4.131, p =0.0002, z=3.590, p=0.002 respectively). Crucially, key rocky intertidal prey species (*P. vulgata, S. balanoides* and *L. saxatilis*) showed similar abundances on the enhanced rock armour to baseline conditions. Cover of algal species such as *F. vesiculosus* and *R. floridula* were recorded in greater abundances in baseline shore platform habitats than on the rock armour revetment (Figure 5-14). *Ulva* (spp.) was recorded in similar levels in both the natural baseline habitat, particularly on plot SP3, and on the rock armour revetment.

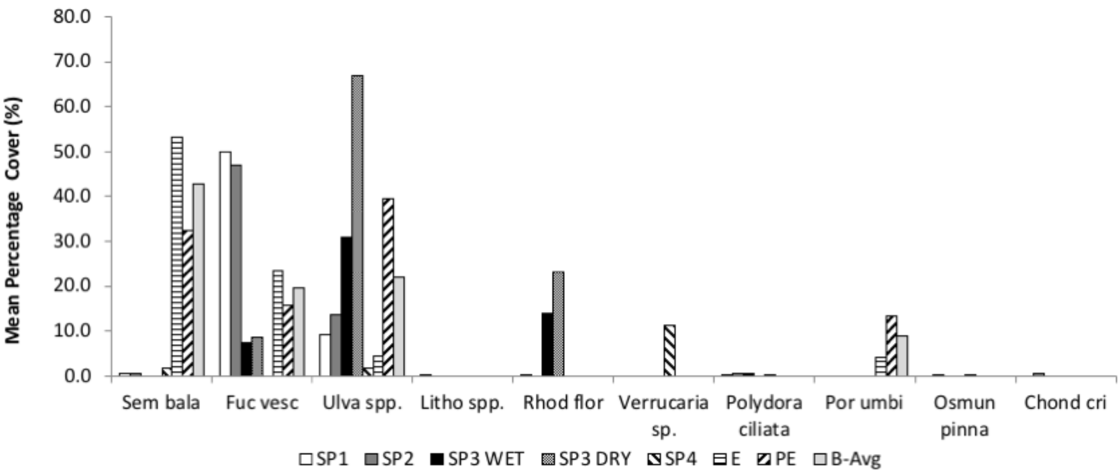


Figure 5-14. Mean percentage cover (%) of algal, lichen and barnacle species within each habitat type. N=5 quadrats for SP1, SP2, SP3 wet and SP3 dry, n=6 for SP4 (where SP=shore platform, as in Figure 5-6) and n=4 for enhanced (E) and partially enhanced (PE) boulders (Naylor et al., 2017b).

5.4.4 October 2016- Mid-upper transect

Two quadrats were taken on each enhanced boulder (one enhanced and one partially enhanced quadrat) and one quadrat on each partially enhanced boulder. Analyses are split between YAI and then enhancement types within YAI. Mean species richness was lower than baseline habitat previously surveyed but did improve slightly between Year 0 (1.53±0.17 SE) and Year 1 (2.97±0.16 SE). Species richness was significantly lower in Year 0 than Year 1 ($H[1]=20.367$, $p<0.001$), with enhancements showing little difference in Year 0 ($H[1]=0.233$, $p=0.629$) but performing better than partially enhanced quadrats in Year 1 ($H[1]=9.111$, $p=0.003$) (Figure 5-15).

P. vulgata was the only mobile species recorded at both sampling sites. No limpets were observed in Year 0 and so Year 1 was found to have a higher abundance of limpets ($H[1]=12.429$, $p=0.0004$), with enhanced quadrats having a greater number of limpets than partially enhanced ($H[1]=16.935$, $p<0.001$). Species richness and abundance is greater on enhanced boulders than partially enhanced boulders in Year 1.

Total number of feature (habitat) types were counted for their occurrence in quadrats but did not influence limpet abundance. This contrasts with previous findings, indicating that quadrat surveys do not fully represent the quantity and cover of passively enhanced habitats and their importance for individual species (features extended beyond the width of the quadrats). Following these surveys, the method was modified to be more representative of rock armour habitat.

S. balanoides, a functionally important habitat forming species, was recorded in significantly greater abundances in Year 1 than Year 0 ($H[1]=13.951$, $p=0.0002$), with only one partially enhanced quadrat in Year 0 recording a 0.5% abundance of barnacles. At this stage, enhancement type made no significant difference within each YAI. Three species of algae were recorded- *F. vesiculosus*, *P. umbilicalis* and *Ulva* (sp.), *Ulva* (sp.) could not be further identified without destructive sampling but the species observed appeared to be *U. intestinalis*. Algae abundance was higher in Year 0 than Year 1 ($H[1]= 7.559$, $p=0.006$), with enhancement type within YAI not exerting any significant impact. The spread of data can be found in Figure 5-15. Figure 5-16 shows that *Ulva* (sp.) is largely responsible for the differences in algae cover between sites ($H[1]=18.514$, $p<0.001$) and the more successionally developed *F. vesiculosus,* absent on Year 0 boulders, was more abundant in Year 1 ($H[1]=7.756$, $p=0.005$). *Ulva* (sp.) was observed in higher abundances on partially enhanced boulders compared to enhanced boulders (Figure 5-16).

The baseline biotope prior to construction at this site within the 0-20 m intertidal extent of the rock armour revetment was classified as that typical of an unstable upper eulittoral rock biotope (LR.FLR.Eph.Ent) (Naylor *et al.*, 2017b; JNCC, 2018). This biotope is characterised by *S. balanoides*, *P. vulgata, Ulva* (spp.) and *F. vesiculosus.* Twelve to eighteen months after installation (Year 1), early colonisation appears similar to the ecological communities replaced within this mid-upper (0-20 m from wall) intertidal extent as characteristic species of this biotope are present. In addition, enhanced boulders look to be performing closer to baseline expectations than partially enhanced boulders (Figure 5-15). As baseline biotope conditions have been met within 24 months, hypothesis 4(c) is accepted.

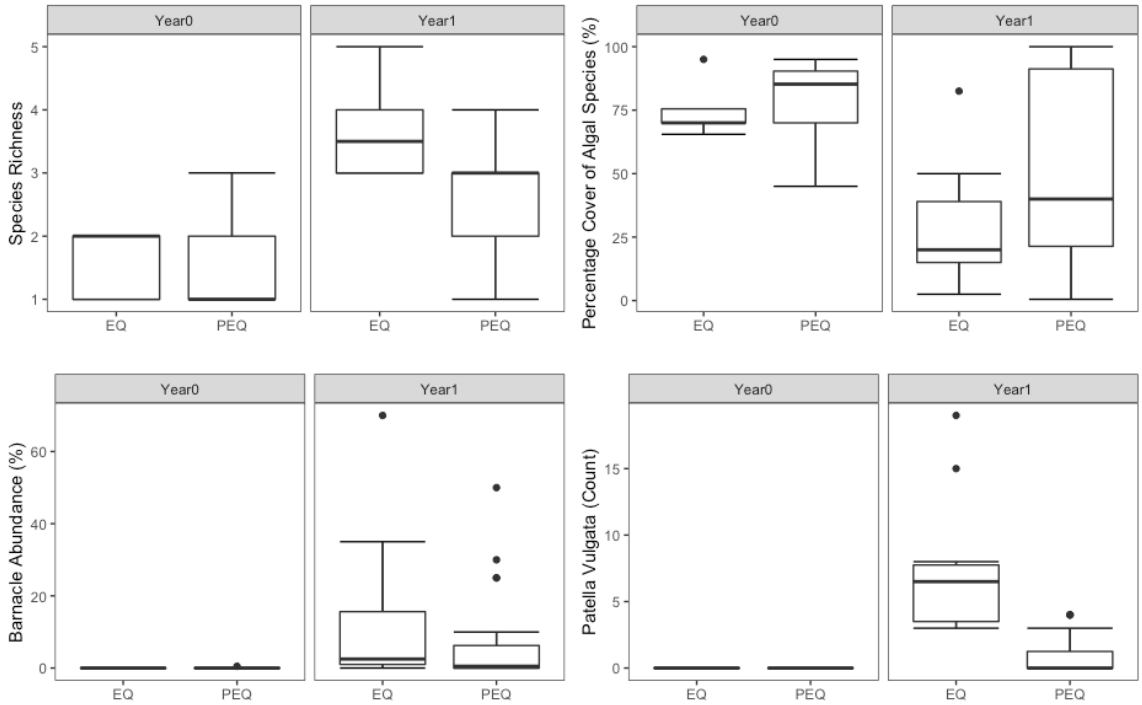


Figure 5-15. Spread of recorded data from enhanced (EQ) and partially enhanced quadrat (PEQ) with years after installation (Year 0 and Year 1). Boxplots show range of data, 25th and 75th percentile and median value for each variable alongside outliers.

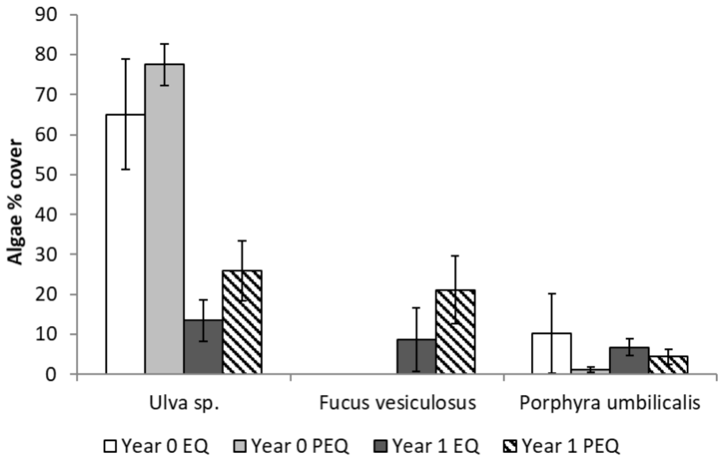


Figure 5-16. Breakdown of algae percentage cover of individual species observed across Year 0 and Year 1 quadrats ($\bar{x}$ ± standard error).

5.4.5 October 2016- Upper transect

At the upper tidal level, Carboniferous limestone was compared to Shap granite to determine differences between lithologies, Carboniferous limestone was only recorded in Year 1 and was sparsely distributed. The upper tidal transect was very species poor, with only *P. umbilicalis* and *Ulva* (sp.) recorded.

In examining the Shap granite boulders, Year 1 had higher species richness than Year 0 (H[1]= 25.865, p<0.001) but no difference was seen with enhancement within either YAI. *P. umbilicalis* was absent from Year 0 quadrats and so had a higher abundance in Year 1 (H[1]=19.646, p<0.001) (Figure 5-17), with enhancements making no difference at this tidal height in either YAI. The same patterns were observed with *Ulva* (sp.) (Figure 5-17) which had a low cover in Year 0 and significantly greater abundance in Year 1 Shap granite boulders (H[1]=15.558, p<0.001).

When comparing lithologies within Year 1, no differences were found between the Shap granite and Carboniferous limestone for *P. umbilicalis* (H[1]= 0.050, p=0.824), *Ulva* (sp.)(H[1]= 1.208, p=0.272) or species richness (H[1]= 0.059259, p=0.8077) (Figure 5-17), therefore hypothesis 1(c) can be rejected.

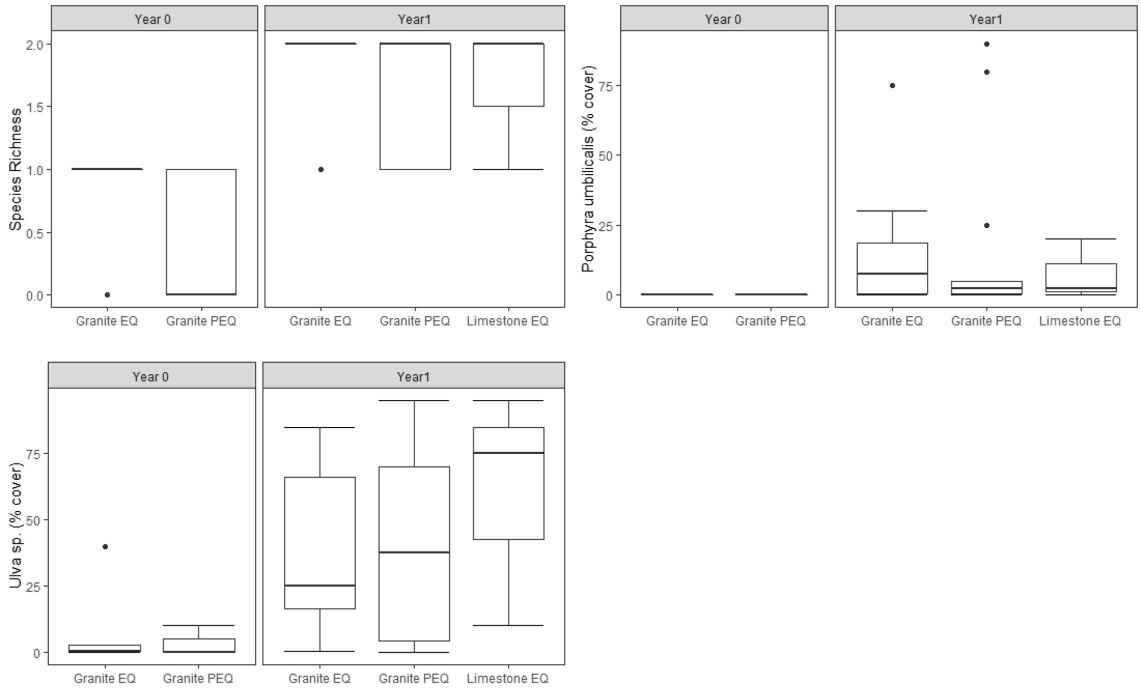


Figure 5-17. Species richness and abundance between YAI and enhancement types in the upper tidal frame. Boxplots show range of data, 25th and 75th percentile and median value for each variable alongside outliers.

5.4.6 June 2017

Baseline richness (September 2016) at Hartlepool found an average of 10 species across shore platform plots, which matches counts of 10 species for enhanced boulders in June 2017 and closely matches 9 species counted on partially enhanced boulders (increased from the six species previously counted in September 2016). Species recorded were consistent across both enhancement types, except of one single *L. saxatilis* on an enhanced boulder (Year 1). The horizontal platform still outperforms the rock armour with a mean species richness of 6.75 (± 0.48 SE) compared to 3.4 species (±0.21 SE) on enhanced boulders and 3.3 species (±0.17 SE) on partially enhanced boulders (when grouped for YAI).

Data collected were analysed for the whole boulder and then subdivided into species found on the surface and found on features in order to identify the most important habitat types for future design consideration. Kruskal-Wallis tests showed that species richness differed between YAI, with Year 0 having more species than Year 1 (z=2.678, p=0.011) and Year 2 exceeding Year 1 (z=-2.654, p=0.012).

As the community developed, species richness increased with time on both the surface of boulders, with Year 2 having greater richness than Year 1 (z=-3.223, p=0.002) and on features, with Year 2 having greater species richness than Year 0 (z=-3.210, p=0.002) and Year 1 (z=-4.826, p<0.001). Species richness was lower on habitat features (e.g. ledges, cracks etc.) than over the surface habitat of the rock (Figure 5-18). Enhancement type did not exert a notable influence on species richness within YAI, likely due to the low average numbers of species found on the rock armour compared to baseline values.

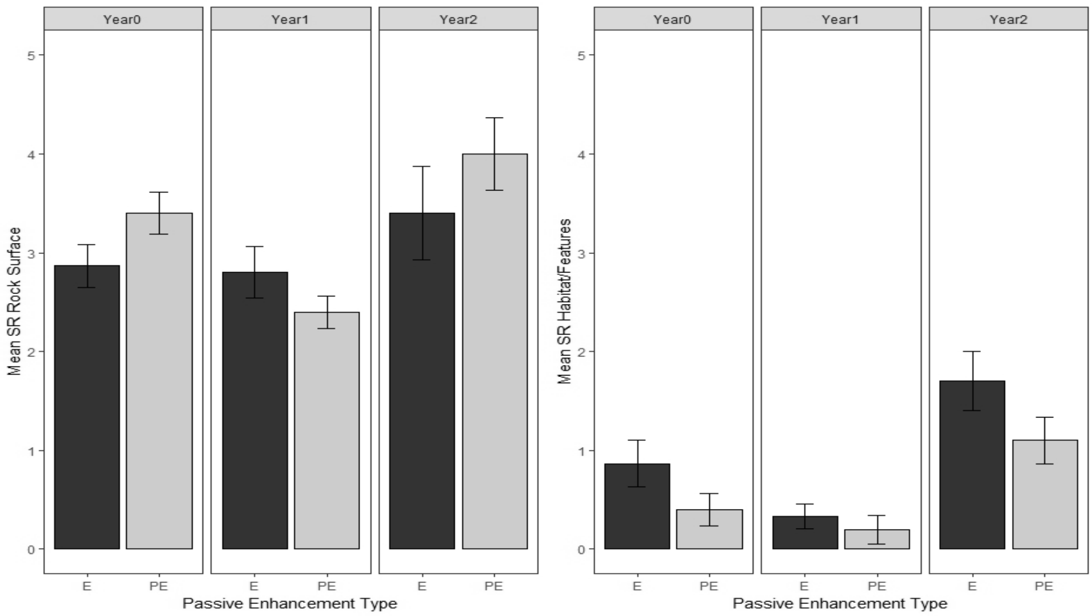


Figure 5-18. Mean species richness for the surface of enhanced and partially enhanced boulders across YAI and for habitat features within boulders across enhancement types and YAI ($\bar{x}$ ± standard error).

Mytilus edulis was recorded on an enhanced (n=1) and partially enhanced (n=1) boulder in Year 0. For most of the mobile species recorded, species were too sparsely distributed for statistical analysis (*L. littorea, L. obtusata, L. saxatilis* and *M. neritoides*) with counts of 2, 7, 1 and 12 respectively across all YAI and enhancement types. The community at Hartlepool was dominated by *P. vulgata*, with enhanced boulders hosting an average of 21.85 (±6.80 SE) limpets compared to 7.78 on partially enhanced boulders (±2.73 SE), with enhanced boulders more in line with the natural baseline for limpet abundance (average of 33.23/m^2). Limpets occurred in significantly greater densities in Year 2 compared to Year 0 (z=-4.414, p<0.001) and Year 1 (z=-7.030, p<0.001) (Figure 5-19). Year 0 also had greater limpet numbers than Year 1 (z=2.925, p=0.005).

Enhancement attracts higher limpet numbers across YAI (z=2.913, p=0.002), with enhanced boulders having 81.5 limpets on average (±16.43 SE) whilst partially enhanced boulders had 27.2 limpets (±8.47 SE). In examining individual years, enhancement made no difference within Year 0 or Year 1 but in Year 2 enhanced boulders had significantly greater limpet abundance than partially enhanced boulders (*H*[1]=8.483, p=0.004). Enhanced boulders promoted a greater number of limpets on both the surface of boulders and on individual features (Figure 5-19). Passively enhanced features make a difference compared to randomly deployed boulders: within Year 2, 51.61% of limpets were found on features compared to 48.39% on the surface as a whole.

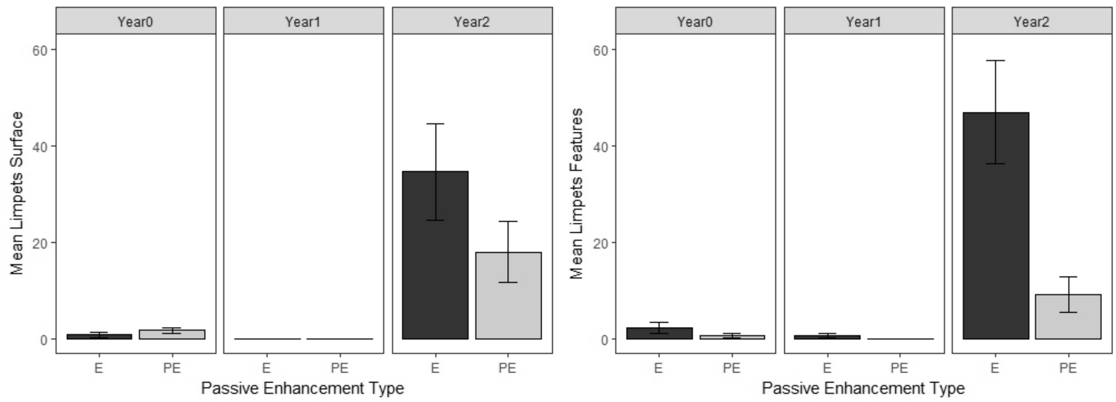


Figure 5-19. Mean limpet abundance on rock surfaces and on features across YAI and enhancement types at Hartlepool (x̅ ± standard error).

In examining the individual habitat types, ledges were the most common features counted across the extent of the rock revetment, with other features occurring in relatively low frequencies (Figure 5-20). This is directly the result of positioning for enhancement of the Shap granite boulders with a higher number of ledges in Year 2. Within the transect areas, blastholes were sporadically distributed and only three were counted on one boulder in Year 0. It is likely that species abundance and richness would increase with a greater diversity of habitat features as water-retaining features, such as pools, are lacking (Figure 5-20).

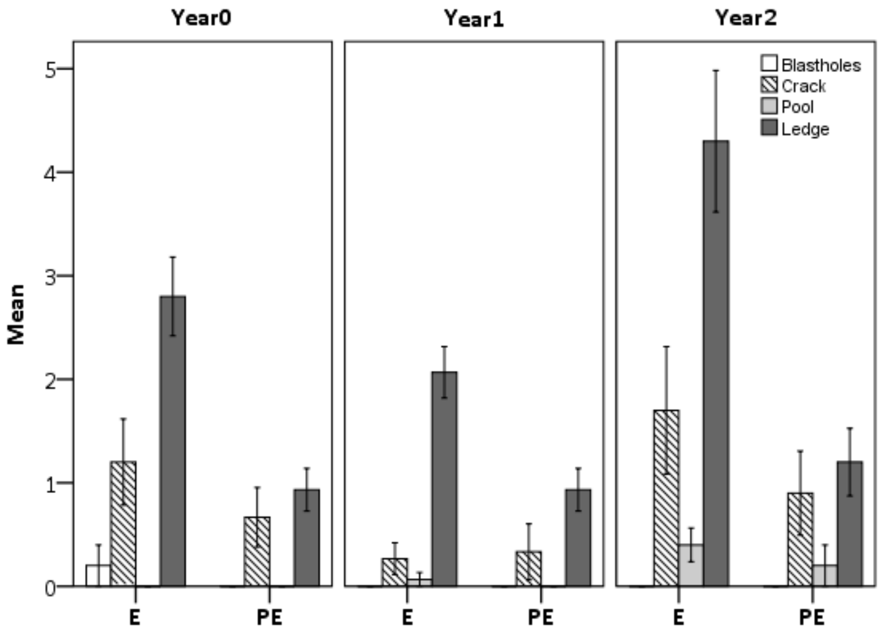


Figure 5-20. Mean number of features on enhanced (E) and partially enhanced (PE) boulders with year after installation (Year 0, Year 1, Year 2) ($\bar{x}$ ± standard error).

Figure 5-21 highlights the significance of feature types across YAI (grouped for enhancement). This aimed to identify the best habitat type and although ledges have greater limpet abundance than other features this is likely due to the higher frequency of ledges within the Shap granite deployed. It is likely that increasing the number of crack (grooves as similar) and pool habitat would increase the amount of limpets as well as other mobile species on the rock armour. As it stands, limpet abundance is greater on enhanced boulders, particularly on larger scale features (dm-10s of dm's) in the form of ledges (Figure 5-21) compared to smaller scale pools, cracks and holes, resulting in the acceptance of hypothesis 3(b).

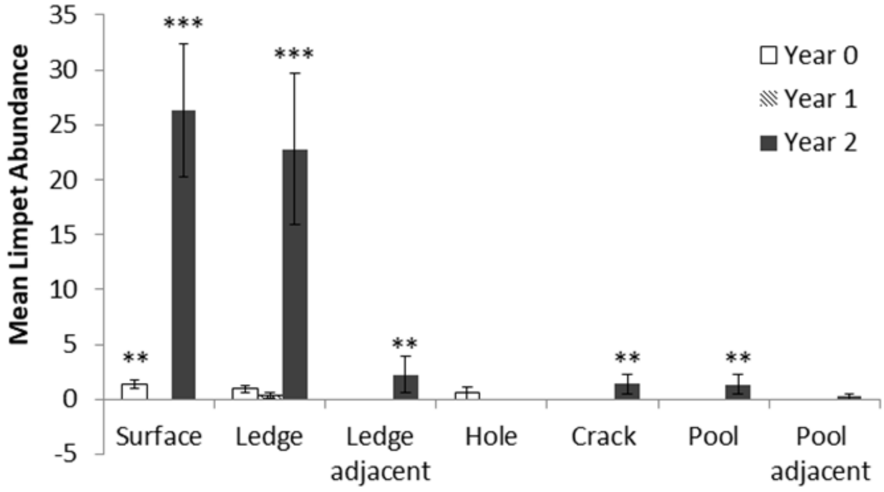


Figure 5-21. Mean limpet abundance on individual features across YAI. Significance is post-hoc Dunn's test results with Bonferroni adjusted p-values (0.05/3=0.016 with p<0.01 ** and p<0.001 ***) ($\overline{x}$ ± standard error).

In examining algal species, *F. vesiculosus, P. umbilicalis, Ulva* (sp.) (likely *Ulva intestinalis*) were observed at the site, with *F. vesiculosus* unique to Year 2, evident of a more successionally developed community. *S. balanoides* did differ by YAI (H[2]= 55.765, p<0.001), with Year 2 having significantly greater barnacle numbers than Year 0 (z=-3.624, p=0.0004) and Year 1 (z=-7.398, p<0.001). Year 0 also showed higher barnacle cover than Year 1 (z=4.220, p<0.001). Within Year 2, enhancement made no difference to barnacle abundance (H[1]=0.052, p=0.819).

Cover of algae decreased in Year 2 compared to other years (Figure 5-22), with both Year 0 (z=3.063, p=0.003) and Year 1 (z=6.021, p<0.001) having greater percentage cover of algae than Year 2. *P. umbilicalis* increased from Year 0 to Year 1 (+48.9%) but then decreased substantially to Year 2 (-88.45% cover), with a similar decrease observed in *Ulva* (sp.) with average percentage cover of 60.92% (Year 0) and 58.5% (Year 1), with cover reducing to 23.02% in Year 2. Enhancement made no difference to algae abundance within Year 2 (H[1]=1.411, p=0.235).

Enhancement made a difference to mobile species abundance but had little influence over species richness and algal species abundance at Hartlepool, resulting in the rejection of hypothesis 4 (a). With the development of the algal community over time and a shift from decreasing *Ulva* (sp.) to increases in other species (like *F. vesiculosus*), hypothesis 4 (b) can be accepted. Additionally, limpet abundance was significantly greater on enhanced than partially enhanced boulder, leading to the acceptance of hypothesis 4 (d).

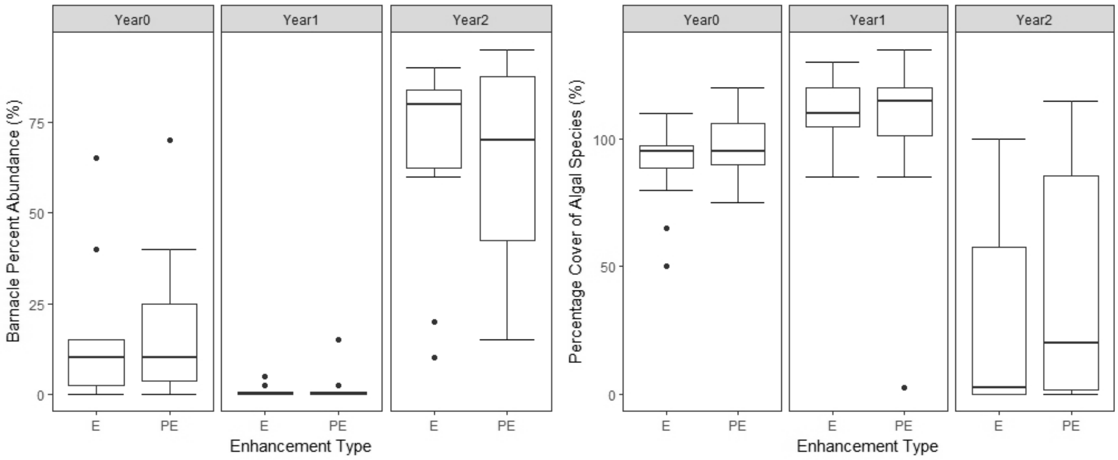


Figure 5-22. Barnacle and algae abundance across YAI and enhancement type. Boxplots show range of data, 25th and 75th percentile and median value for each variable alongside outliers.

5.4.7 September 2017

5.4.7.1 Upper tidal level

Average species richness was limited for both Shap granite enhanced (1.73 ±0.15 SE) and partially enhanced boulders (1.8 ± 0.14 SE) and limestone boulders (1.53 ± 0.13 SE) surveyed at the upper tidal level of Year 2. No differences were found between enhancement types (enhanced, partially enhanced and limestone) or when comparing the Shap granite (enhancements grouped) to Carboniferous limestone ($H[1]=3.567$, p=0.059). *Ulva* (sp.) and *P. umbilicalis* were the only two species observed at this tidal height. *Ulva* (sp.) abundance did not vary between lithologies or between enhancement types. *P. umbilicalis* differed between enhancement types with enhanced boulders (22% ± 7.15 SE) having a higher percentage cover than limestone (4.7% ± 2.62 SE) (z=2.663, p=0.012). Partially enhanced boulders also had a greater abundance (12.5% ± 3.8 SE) than limestone but this was not significant. Combining enhancements confirmed that granite had a greater cover of *P. umbilicalis* than limestone at the upper tidal level (z=2.916, p=0.002).

5.4.7.2 Hartlepool and Skinningrove- rock material field comparison

This comparison is conducted within the Year 2 section of the rock revetment at Hartlepool and the comparably aged Skinningrove installation. At Hartlepool, six species were recorded on the rock armour, with a single observation of *N. lapillus* on a partially enhanced boulder. In contrast, eleven species were recorded on the rock armour at Skinningrove. Skinningrove had an average of 5.40 (±0.27 SE) species compared to 3.33 (±0.24 SE) at Hartlepool, making Skinningrove significantly more species rich ($z= 4.672$, $p<0.001$). Enhancement did not influence species richness at Hartlepool ($H[1]=1.260$, $p=0.262$), likely due to the rock armour being relatively species-poor (3.07 ±0.33 SE for enhanced and 3.60 ±0.34 SE for partially enhanced).

Only two mobile species were identified at Hartlepool, compared to four at Skinningrove. Although *P. vulgata* was the most abundant species at both sites, low numbers of *L.littorea* (n=1)*, Talitrus saltator* (n=30 across several boulders) and *L. oceanica* (n=1) were observed at Skinningrove. Due to the low abundance and sparse distribution of other mobile species, analysis focused on limpet abundance. The same approach was adopted for Hartlepool.

Figure 5-23 highlights the difference in limpet abundance at both sites, with significantly greater limpet numbers observed at Hartlepool than Skinningrove ($H[1]=23.482$, $p<0.001$). Hartlepool is a limpet dominated site, with significantly greater numbers of limpets on the surface of boulders ($z=4.456$, $p<0.001$) and on individual habitat features ($z=3.296$, $p<0.001$) than Skinningrove. Although limpet abundance was greater at Hartlepool, Skinningrove had significantly greater species richness, leading to the rejection of hypothesis 1(b), that Shap granite performs better than the smoother, darker Norwegian granite in terms of species richness and abundance.

In examining limpet abundance on the dm-scale ledges, Hartlepool had more limpets on ledges than Skinningrove ($z=3.923$, $p<0.001$), with more limpets on ledges on enhanced boulders compared to partially enhanced boulders ($H[1]=9.130$, $p=0.003$). This difference is likely as boulders at Skinningrove were not oriented to be ‘passively enhanced’. As ledge habitat has been identified as important from previous studies within this chapter, this suggests that passive enhancement does exert some influence in attracting important prey species.

S. balanoides abundance differed between sites, with Hartlepool having a greater percentage cover of barnacles than Skinningrove (z=6.254, p<0.001) (Figure 5-23). Laboratory tests showed that roughness did not differ between the Shap granite and Norwegian granite, indicating that hypothesis 3(a), that early colonising barnacle abundance is unrelated to roughness in Shap granite or Norwegian granite can be accepted.

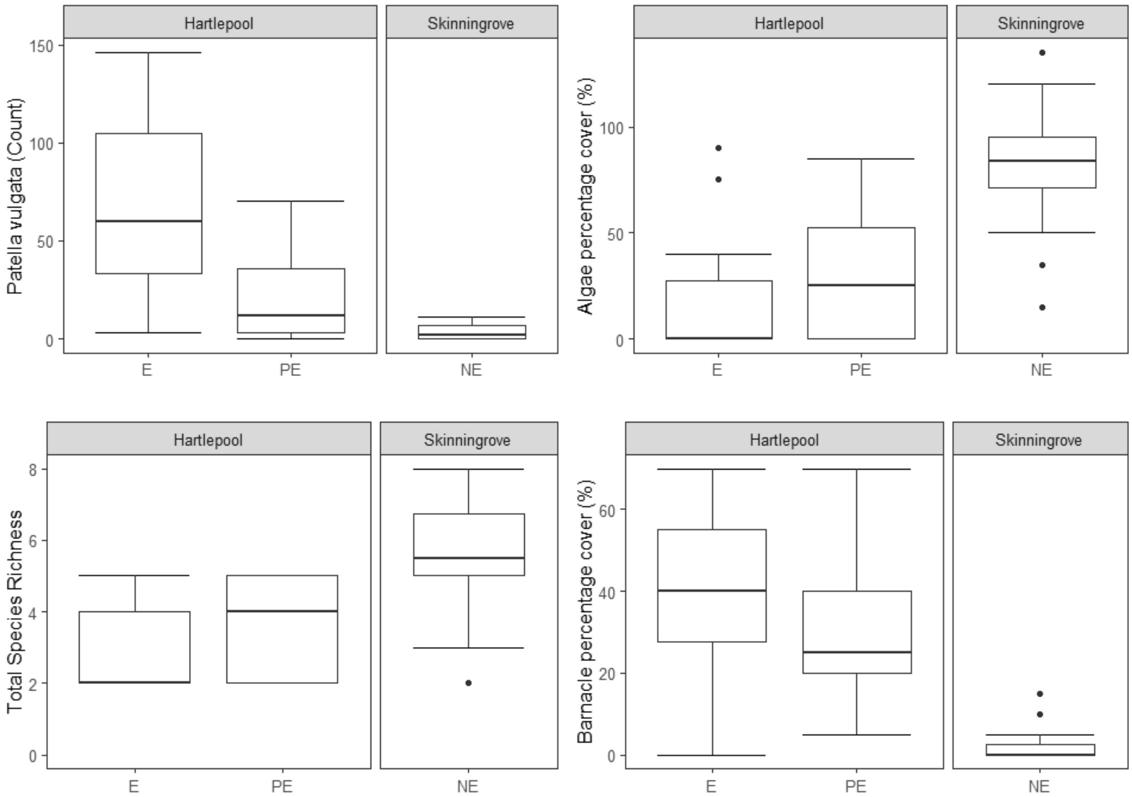


Figure 5-23. Limpet counts, algae and lichen abundance, species richness and barnacle abundance at Hartlepool and Skinningrove (2 YAI) between enhancement types (E=enhanced, PE= partially enhanced, NE= non enhanced). Boxplots show range of data, 25th and 75th percentile and median value for each variable alongside outliers.

At Skinningrove, *Ulva* (spp.), *F. vesiculosus* and *F. spiralis* were recorded in the highest abundances, with *C. crispus* and *P. umbilicalis* also recorded in low frequencies (Figure 5-24). Hartlepool had significantly less algae cover than Skinningrove (z=-5.680, p<0.001) but the diversity of algae species at Skinningrove suggest it is a more successionally advanced community.

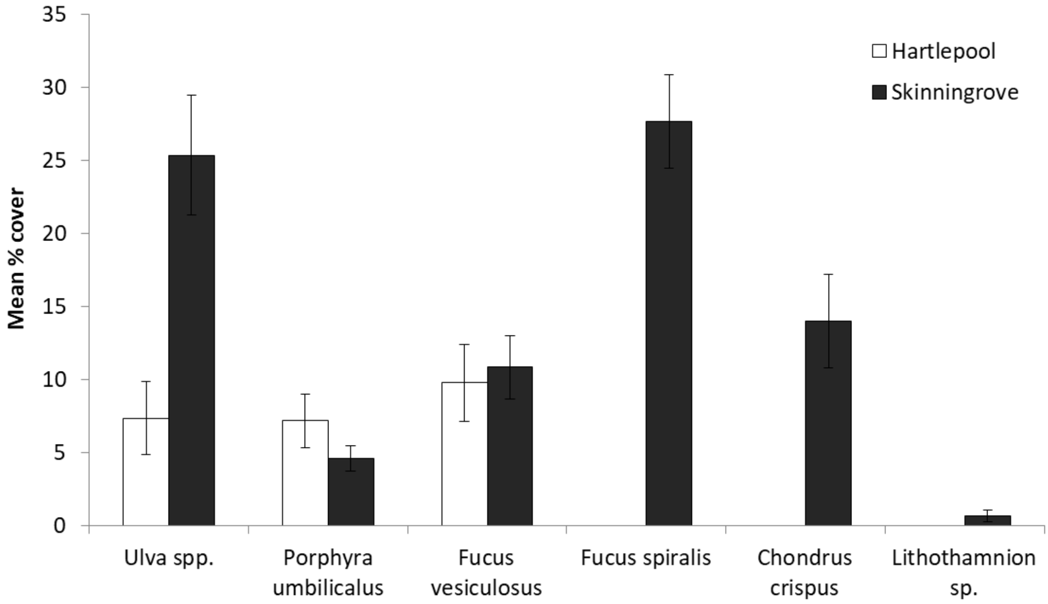


Figure 5-24. Mean percentage cover (%) of individual species of algae identified at Hartlepool and Skinningrove (±SE) ($\bar{x}$ ± standard error).

5.4.7.3 Quarried features- Hartlepool and Skinningrove

At Hartlepool, there was no significant difference between the presence of blast holes on boulders and the species richness of adjacent boulders. At Skinningrove, enhanced boulders with blast lines had significantly greater species richness than adjacent boulders ($H[1]=5.503$, p=0.019). Since limpets were the primary mobile species, they were used for statistical analyses. At Skinningrove, limpets were more abundant on boulders enhanced by quarried features such as blast holes than adjacent boulders ($H[1]= 12.376$, p=0.0004) (Figure 5-25), yet the presence of limpets within these features was not significant ($H[1]=1.055$, p=0.304). At Hartlepool, the sporadic and rare occurrence of blast features (Figure 5-25) appears to have made little difference to limpet abundance.

Skinningrove had many blast lines on many boulders. In examining boulders with quarried blast features, 51.75% of limpets sampled at Skinningrove were found inside these blast features compared to 29.82% on the boulder surface. Blast lines at Skinningrove were between 1.5-2 cm deep but were not optimally positioned and did not hold water. At Hartlepool, 34.91% of limpets observed resided in the blast holes compared to 45.26% on the boulder surface. However, the Hartlepool blast holes were open at the bottom allowing water to vacate. Differences in species richness with blast lines and adjacent boulders at Skinningrove are unlikely to be solely the result of the blast features. This combined with the lack of influence of these features on species richness at Hartlepool and the insignificant numbers of limpets present within features leads to the rejection of hypothesis 5(a), that quarried features on boulders would host greater species richness and abundance than adjacent surfaces.

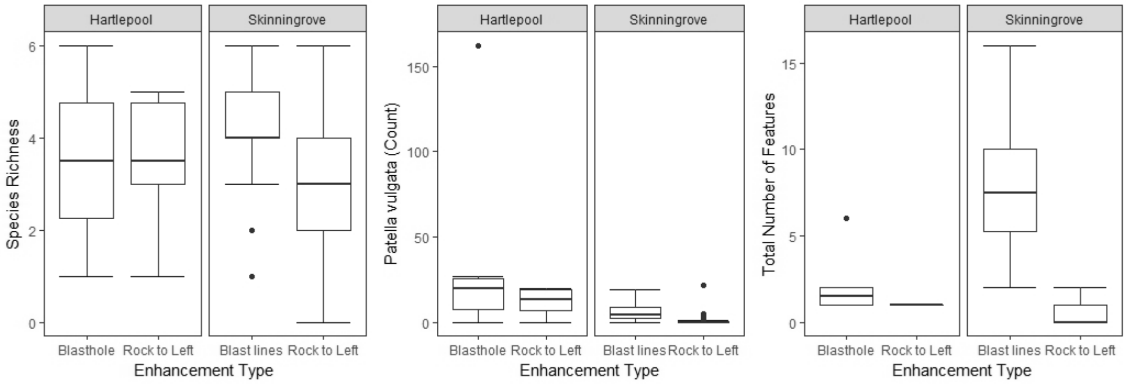


Figure 5-25. Counts of species richness, *P. vulgata* and total number of features on boulders with quarried features and boulders to their left.

In comparing the main feature at each site, blast lines at Skinningrove had very little influence (cm-scale variation) on the surface roughness of the boulder compared to the dm-scale features at Hartlepool (Figure 5-26). Improving the positioning of these features and their depth may increase their effectiveness as habitat as the dm-scale ledges at Hartlepool were shown to influence mobile species abundance.

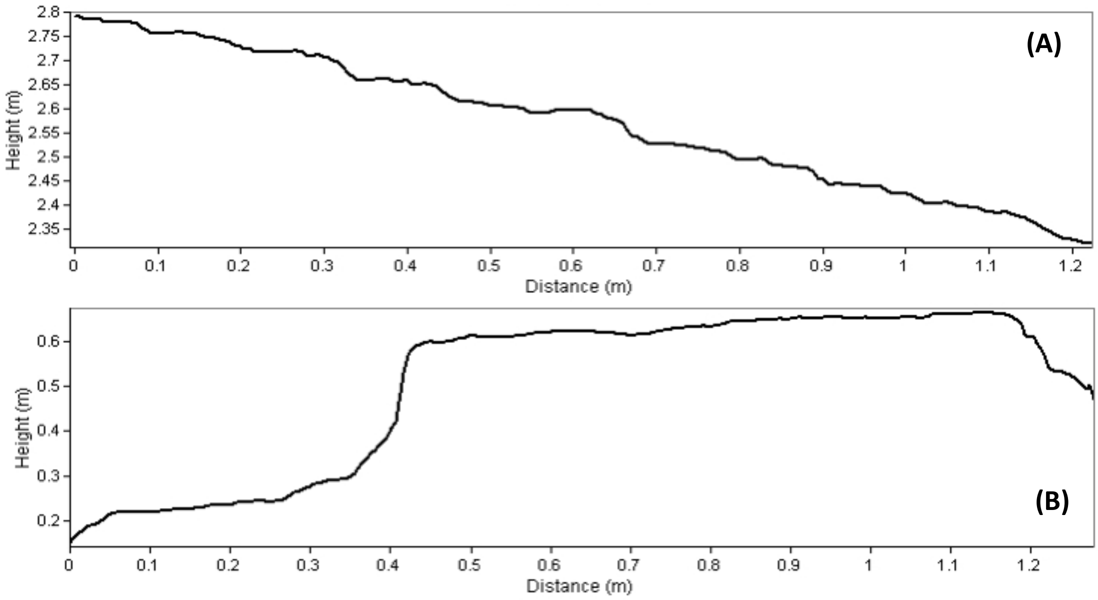


Figure 5-26. Surface profiles of boulder features for (A) Skinningrove 'blastlines' and (B) Hartlepool 'ledge' habitat.

5.5 Summary

Table 5-9 is a summary of the original hypotheses (Section 5.2.1) and shows that passive enhancement strategies at the dm-scale are effective in increasing the abundance of key intertidal species. These enhancements aided in matching baseline conditions within 18 months and in combination with more ecologically favourable lithologies (lighter coloured, more porous lithologies), may further increase ecological potential in future.

Table 5-9. Summary of hypotheses and reasoning for accepting (✓) or rejecting (✗) each.

	Hypothesis	Conclusion	Explanation
1a	Lithologies with a combination of lighter colour, higher surface roughness and higher porosity will have greater ecological suitability.	✓	After ranking rock material properties (Table 5-10), this can be confirmed.
1b	Shap granite performs better than the smoother, darker Norwegian granite in terms of species richness and abundance.	✗	Shap granite had greater mobile species abundance due to high limpet densities but did not match the Norwegian granite in terms of species richness.
1c	Carboniferous limestone has greater species richness and abundance than Shap granite.	✗	The positioning of the Carboniferous limestone within the upper tidal zone resulted in it not being optimally positioned to take advantage of its material properties. Limited species richness and abundance was recorded at this tidal height so no significant differences occurred between lithologies.
2a	Natural surfaces have greater species richness and abundance than artificial surfaces in baseline surveys.	✗	Enhanced boulders matched natural shore platforms for species richness and the abundance of certain species.
3a	Mobile species abundance is greater on larger scale (dm-10s of dm's) features than smaller scale features (cm-dm).	✓	Ledges are shown to be an effective habitat in increasing mobile species abundance compared to features such as cracks and pools. However, these smaller scale features were sporadically recorded and so ledges were the dominant feature.

3b	Early colonising barnacle abundance is unrelated to roughness in Shap granite or Norwegian granite.	✓	Barnacle abundance is significantly greater on Shap granite at Hartlepool than the Norwegian granite at Skinningrove. This is likely the result of closer distance to larval supply (adjacent platform versus sandy shore at Skinningrove).
4a	Species richness and mobile species abundance is greater and algal abundance lower on enhanced boulders than on partially enhanced	✗	Enhancement influenced mobile species abundance but had little influence over species richness and algal species abundance at Hartlepool
4b	Enhanced boulders are more successionally advanced after 24 months than partially enhanced boulders.	✓	Algae community shifting from *Ulva* sp. to *F. vesiculosus* and grazers, suggesting successional development.
4c	Enhanced boulders match baseline biotope conditions within 24 months.	✓	Enhanced boulders matched baseline biotope conditions, characterised by *S. balanoides*, *P. vulgata, Ulva* (spp.) and *F. vesiculosus,* within 18 months.
4d	Abundance of key prey species (gastropods) for internationally important waterbirds is greater on enhanced boulders than on partially enhanced boulders.	✓	Limpet abundance was significantly greater on enhanced than partially enhanced boulders during surveys in June and September 2017.
5a	Quarried features on boulders host greater species richness and abundance than adjacent surfaces.	✗	Quarried features at Hartlepool were sporadically distributed and rare in occurrence and at Skinningrove were not optimally positioned to maximise ecological potential. As such, these features made little difference to species richness and abundance.

5.6 Discussion

The laboratory tests on rock material properties offer an insight into the ecological suitability of materials at the mm-cm scale. Previous work has shown granite to be less ecologically suitable than limestone due to a higher likelihood of darker rock surfaces reaching lethal temperatures for organisms and have lower potential for biological weathering to increase surface roughness (Coombes et al., 2011; Coombes and Naylor, 2012). In selecting a rock material for a revetment, albedo is an important factor since colonising species have different thermal tolerances, with darker lithologies likely to increase thermal and desiccation stress (Coombes and Naylor, 2012; Meager et al., 2011). This is particularly important in the context of any predicted warming under climate change and optimising material choice to select lithologies with high dry albedos will aid survival of intertidal species (Kordas et al., 2014). Similarly the surfaces of lithologies with high porosity and water absorption capacity remain wetter for longer, favouring the survival of intertidal species during the low tide period.

The biogeomorphological potential, i.e. the ability of species to erode the substrate through boring or grazing (biogemorphic alternation), and rock hardness also impacts its long term surface roughness. These factors increase the habitat complexity of the rock by increasing the potential for microhabitat creation and subsequent availability to provide refuge and retain water, further reducing the impacts of intertidal stressors (Firth et al., 2013; Sherrard et al., 2016).

In theory, the above argument would result in the Norwegian granite at Skinningrove being less ecologically suitable than the Shap granite at Hartlepool due to its darker colour. However, although mobile abundance was greater at Hartlepool, species richness was greater on the Norwegian granite. It is likely that context is key (Green et al., 2012) and local climate, wave exposure and larval supply has likely influenced the variation in species richness between the two field sites (Menge et al., 1997; Moschella et al., 2005; Sherrard et al., 2016). This is also true of the positioning of the Carboniferous limestone at Hartlepool, which was expected to be more ecologically suitable than the Shap granite. However, it was too expensive to deploy the Carboniferous limestone throughout the rock armour revetment and it was only sporadically placed in the upper tidal levels, making it poorly sited for optimising ecological potential as few species colonise at this level.

Passive positioning of boulders served to increase the abundance of *P. vulgata* on the rock armour revetment at Hartlepool compared with control, randomly deployed partially enhanced boulders. Limpet abundance made a significant difference to colonisation patterns within 2 years and were found to congregate on ledges compared to other available habitat types. Orientation for ledges increased habitat complexity at the cm-dm's scale by increasing the surface area available (Loke and Todd, 2016) for mobile species to colonise. High wave energy can increase the frequency of dislodgement or interrupt limpet foraging (Denny, 1985) but it is likely that the angled, vertical faces of ledges favoured limpets colonisation by locally reducing wave impact. However, the lack or low occurrence of other habitat features at Hartlepool temper the assertion that limpets favour ledges as a result of passive positioning.

Despite boulder placement for specific habitat features, species richness was not influenced by this level of passive enhancement at Hartlepool and did not match baseline values on the adjacent shore platform. This could be a result of a lack of suitable habitat features, such as the pools that were widely available on the Magnesian limestone platform or that the Magnesian limestone has more suitable rock material properties. It is also possible that the wave exposed environment, combined with heavy disturbance of the adjacent natural rocky shore by construction vehicles, disrupted source populations and impacted the initial recruitment of species onto the recently installed coastal defences (Blanchard and Bourget, 1999; Evans et al., 2015; Schoch and Dethier, 1996). With construction now ended, following September 2017 surveys, the reduction in anthropogenic disturbance may result in a larger number of species colonising the rock armour (Benedetti-Cecchi et al., 2003; Lee and Sin, 2009).

Another factor which may account for the limited effect of enhancements on species richness is the habitat suitability post-recruitment. Despite passive orientation, there was a limited variety of microhabitats available, reducing the availability of potential niches. A lack of microhabitats, particularly those that retain water, limits the potential for lower-shore and desiccation-sensitive species to colonise the more physically stressful boulders located in the mid-upper intertidal zones (Firth et al., 2013; Moschella et al., 2005; Pister, 2009). Although recommendations were made to increase the number of water-retaining features at Hartlepool, orientation for surface depressions was not at a sufficient level of enhancement to increase water retention on the surface of rock armour, with fewer features and when present held no more than 1.5 cm of water. This limited the ability of these 'pools' to match levels of species richness typically found in rock pools (Evans et al., 2015; Jackson, 2015). As artificial structures at this site have been consistently shown to be relatively species-poor compared to the natural platform they replaced, it is fundamental that future rock armour installations maximise the variety of habitat types available or retrofit pools to structures to increase the number of water-retaining features and subsequently increase the diversity of colonising species (Browne and Chapman, 2011; Firth et al., 2016a), potentially to baseline levels.

Although the Shap granite is not the best lithology for ecology, Hartlepool is also a species-poor location and so the recommendation of this lithology may well have been the best of the options available. The surface roughness of the Shap granite meant that it had high mm-cm and cm-dm's scale roughness whereas the Norwegian granite lacked this higher level of roughness and did not have the larger features such as ledges as widely distributed on the boulder surfaces. Therefore, Shap granite remains the optimal of the two granites for passive enhancement for ecological gains.

Previous studies involving retrofitted active enhancements of grooves and pits, found that grooves and pits performed similarly but features with greater water holding capacity resulted in a greater number of species observed (Firth et al., 2014b; Hall et al., 2018). The examined quarried blastholes and blastlines, which were accidental enhancements at the cm-scale, were expected to perform similarly to active enhancements at a similar scale that have been shown to provide favourable ecological results (Firth et al., 2014b; Hall et al., 2018). However, quarried features performed poorly and although limpets were more abundant on quarried boulders at Skinningrove, limpet presence within these features was insignificant. Grooves can provide an important microhabitat for species (Borsje et al., 2011) but the blastlines at Skinningrove were not optimally positioned to retain water on the surface and many were too shallow (approximately 1.5 cm deep) or positioned too high in the tidal frame for species colonisation to provide a high level of habitat provision. In addition, the bottoms of blastholes at Hartlepool were open and had no water retention. If these features were in-filled to a desirable depth, they could function similarly to rock pools and provide water retaining features. Although the results of this study were not statistically significant, the scale and orientation such features can function as effective habitat on rock armour revetments. Sympathetic positioning of blast features may have allowed the above results to approach the increased species diversity and abundance typically found in active enhancement studies with pits and grooves (Firth et al., 2014b; Hall et al., 2018).

Barnacle abundance was significantly greater at Hartlepool than at Skinningrove, which cannot be attributed to differences in mm-scale roughness as these were shown to be statistically equal. This difference may be a result of biogeographic conditions and larval supply (Menge et al., 2010) as Skinningrove was fronted by a sandy beach while Hartlepool had an immediate larvae source as it was fronted by a natural rocky shore platform.

5.6.1 Ecological suitability – engineering recommendations

Both laboratory and field experiments allow an ecological suitability ranking for each lithology to be presented (Table 5-10), including two additional lithologies (Cornish granite and Portland limestone) used in previous studies and employing similar methods, inserted here for comparative purposes only (Coombes, 2011; Coombes and Naylor, 2012). Estimates of rock density were made from CIRIA (2007) and the grades of rock hardness established from Hoek and Brown (1997) methods were converted into the low, moderate and high categories in Table 5-10. Hardness was scored according to ecological suitability, with lower hardness lithologies scoring higher in this category. Ecological engineering potential was estimated from combining understanding of calcium content and porosity/WAC and from previous studies (Coombes, 2011; Coombes and Naylor, 2012).

In examining rock materials in the laboratory, the Magnesian limestone and the Portland limestone are the most ecologically suitable materials as they combine the most desirable features of being softer lithologies (more suitable for habitat creation and boring species, Figure 5-27) with high mm-scale surface roughness (encourages early stage colonisation), high albedo (reduces thermal stress for intertidal species) and high water absorption capacity and porosity (more holes and potential for habitat creation and retains water, reducing desiccation stress). These two lithologies also had the highest biogeomorphological potential, where boring and grazing species would be capable of modifying the substrate and further increasing surface roughness over time, as seen in the 103.93% increase in roughness between rough and weathered samples of Magnesian limestone (Section 5.4.1, Figure 5-27). Although not tested, the limestones have higher calcium content than the examined granites, which has the potential to exert a positive influence on community composition and diversity (Chapter 2.2).

Figure 5-27. Magnesian limestone platform surface at Hartlepool showing bioerosion by limpets (limpet homescars and resulting pits).

At the scale of laboratory testing, the Portland limestone offers the balance of having the characteristics that make it ecologically suitable and since it can be blasted, it is more suitable than the Magnesian limestone for engineering purposes. The Carboniferous limestone at Hartlepool was suitable for both ecology and engineering purposes as it has moderate fine-scale (mm-scale) surface roughness and albedo. Additionally, its low porosity and water absorption capacity increases its resistance to abrasion and weathering and makes it more suitable for engineering schemes that require durability (French, 2001; Crossman et al., 2003). The Welsh Carboniferous limestone matched the suitable albedo and low porosity values of the Hartlepool Carboniferous limestone but had higher fine-scale roughness, which would benefit early colonising species such as barnacles. The moderate hardness of this lithology also increases its engineering suitability. The Blue Lias limestone performed similarly to the Carboniferous (Hartlepool) limestone at the scale of the laboratory tests (Table 5-10) but had a lower albedo, slightly reducing its ecological suitability. Additionally, the Blue Lias has limited engineering suitability as it has thin horizontal beds that render it unsuitable for blasting and coastal engineering applications.

Rock material properties are key to providing a base level of enhancement that increases the potential for early colonisation of species as well as contributing to the reduction of thermal and desiccation stress that these organisms may undergo in the intertidal zone. Incorporating the larger scale rock mass features (cm-dm’s scale) identified in field studies to the analysis of ecological suitability notably increases the suitability of the lithologies tested (Table 5-10). Larger scale surface features, particularly ledges (dm-scale) in this study, increased mobile species abundance and so to fully passively optimise for intertidal ecology, both rock mass and rock material properties must be considered together. However, further field testing is needed to incorporate other lithologies used in coastal engineering (absent from Table 5-10). These results would also benefit from cm-scale roughness measurements as this scale is absent from the current analysis but is important for ecology as it increases available surface area for attachment.

The recommended lithologies from laboratory testing and field testing are different, highlighting the importance of combining findings of the rock mass and rock material property comparisons. From laboratory testing, the recommended lithologies that combine ecological and engineering benefits are the Portland limestone and the two Carboniferous limestones (Welsh and Hartlepool samples). Incorporating cm-dm scale features result in these recommendations shifting to the Shap and Norwegian granites and the Carboniferous limestone at Hartlepool but these results are more limited as only several lithologies were surveyed or observed in the field. If the design considered deploying additional lithologies that would function purely as ecological hotspots within the structures, then the Magnesian limestone would provide this.

Table 5-10. Ecological suitability ranking (1=Low (L), 2=Moderate (M), 3= High (H)) for rock material and rock mass properties from laboratory and field experiments on unweathered rock samples. SG= Shap granite, NG=Norwegian granite, CL= Carboniferous limestone (H)= Hartlepool, (W)=Welsh, ML=Magnesian limestone, BL= Blue Lias limestone, PL=Portland limestone, CG=Cornish granite. Portland limestone and Cornish granite from Coombes and Naylor (2012) and Coombes (2011). Blank space indicates data not collected.

	Lithology							
	Igneous			**Sedimentary**				
	SG	NG	CG	CL (H)	CL (W)	ML	BL	PL
Calcium Content	L	L	L	H	H	H	H	H
Hardness	H	H	H	L	M	L	L	L
Density	M	M	M	M	M	M	M	M
Albedo	M	L	M	M	M	M	L	M
Porosity	L	L	L	L	L	H	L	H
WAC	L	L	L	L	L	H	L	H
Surface Roughness (25 mm^2 scale)	H	H	H	M	H	L	M	H
Long-term biogeomorphological potential	L	L	L	M	M	H	M	H
Ecological suitability (mm-cm scale) [Lab tests]	**12**	**11**	**12**	**16**	**16**	**20**	**15**	**22**
Ledge habitat (Field tests)	H	M		M		H		
Pool habitat (Field tests)	L	L		L		H		
Blast features (Field tests)	L	H		L		L		
Ecological suitability with cm-dm's scale features [Lab + Field]	**17**	**17**		**21**		**27**		
Engineering suitability	H	H	H	H	H	L	L	M

5.7 Conclusion

The passive positioning of boulders at Hartlepool satisfies the initial aims of the scheme, with baseline abundance of key waterbird prey (*Patella vulgata*) matching baseline values within 24 months. At the Hartlepool site, the passive enhancement of large-scale features (decimetres scale) is optimal for increasing mobile abundance and in order to increase species richness, post-quarrying improvements to existing quarried features could perform similarly to active enhancements if designed properly. This combination would contribute to maximising biodiversity potential on similar structures.

The findings from the laboratory and field tests examining rock mass and material properties suggest that early colonisers (e.g. barnacles) and ecological potential are favoured by rock armour revetments incorporating a light-coloured lithology with high surface roughness from the mm-dm scale, either through naturally occurring or quarried features to maximise multi-scale surface roughness, that has long-term biogeomorphological potential.

Passive enhancement would be best suited alongside active enhancement to reduce the overall cost of enhancements whilst increasing the habitat potential. These suggested active enhancements include the deliberate modification of existing features, such as sealing quarried blastholes to trap water and the better positioning of boulders with existing blastlines, and where required, adding a lip with concrete to blastlines so that there is greater water holding potential. This in turn will likely increase the recruitment of specific species and increase species richness. Passive enhancements on their own are not enough to maximise species richness on the structure or create an ecological gain but it is better than doing nothing at all.

This study demonstrates the need for maximising habitat potential on artificial structures and the need to consider both the physical scale and orientation of surface features when designing for ecology. It offers a low-cost strategy for enhancing rock armour that requires little expertise to be implemented and biodiversity gains maximised. Passive material choice is a crucial first step in determining ecological suitability and with optimal positioning for macro-scale features such as ledges there is a good minimum standard for ecologically enhanced engineering practice. There is a need for a full-scale trial of rock armour with different lithologies side by side to fully address the question of identifying optimal lithology and features for maximising ecological potential on rock armour revetments. Key to this is interdisciplinary collaboration between coastal scientists, engineers and contractors.

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Chapter 6. Synthesis - Interactions between coastal biodiversity and geodiversity: informing engineered design

6.1 Research summary and main findings

Topographic and habitat complexity are both important in maintaining the diversity and abundance of ecological communities (Kostylev et al., 2005; MacArthur and MacArthur, 1961; Meager et al., 2011; Pinn et al., 2008). With increased artificial modification of the coastline globally (Bulleri and Chapman, 2010; Gacia et al., 2007) together with increased pressure on ecosystems from the combined effects of urbanisation and climate change (Hall et al., 2018), including sea level rise and increasing storminess, there is a need to create or replace habitat aimed at increasing biodiversity on artificial structures.

The research reported here is a multi-scale (mm-km's) and multi-site approach to improve the understanding of the interactions between geodiversity and biodiversity on intertidal rocky shores. Aiming to identify the influence exerted by rock material (lithology, albedo, porosity, roughness), rock mass (structure) properties and associated geomorphic features (microhabitats) on rocky intertidal species, these relationships are used to assess the effectiveness of ecological enhancement strategies on artificial structures. This has been addressed on natural shores through the development of traditional ecological survey techniques via quadrat sampling to better account for lithology, microhabitats and their associated dimensions. Designs for the enhancement of artificial structures by retrofitting tiles of varying complexity on vertical coastal infrastructure and the passive positioning and optimised material choice of rock armour boulders have been monitored to assess the efficacy of any future ecological enhancement strategies.

Chapter 2 established that at a coarse scale of 100s of km, the effects of lithology did not contribute to substantial variations in biodiversity but the influence of rock mass structure and associated geomorphic features was not examined (Figure 6-1). The results indicated that community composition was consistently similar between contrasting lithologies within regions (east and west coast of Scotland and south Wales). Significant results, such as the greater mobile species richness and abundance found on sedimentary lithologies over metamorphic lithologies at the national scale, were likely due to variation between locations including wave exposure, water temperature (Schoch and Dethier, 1996) and planktonic food supply (Chapter 2.6). This chapter determined that the effects of lithology may be more prominent at smaller spatial scales than the broad scale analysis conducted in Chapter 2 and noted the importance of including rock mass properties in better understanding the influence of geodiversity on biodiversity.

Chapter 3 examined the influence of rock material (lithology) and rock mass (structural habitat features) properties within and between four sites selected from those surveyed in Chapter 2 alongside two other separate sites. Geology (rock mass and material properties of rocks) and geomorphology exerted more of an influence when comparing between sites (10s of km) as the presence and abundance of individual microhabitats were found to vary with lithology. For example, the natural limestone shore at Girvan was found to be the most complex due to the mm-cm scale variability in surface roughness (personal observation) alongside the high abundance of cm-dm+ scale features including deep pools and ledges. This mm-cm variability is likely a result of the limestone being softer, reducing its resistance to erosion and weathering processes and facilitating intertidal organisms contributing to geomorphology through bioerosion, increasing the overall surface complexity of these softer lithologies (Coombes et al., 2011; Firth et al., 2012; Spencer and Viles, 2002; Vidal et al., 2013). Geomorphic features within natural shore platforms vary by lithology which in turn exerts an influence over the biota (Figure 6-1). Although the exact influence of features is highly variable with location, an overarching pattern was found whereby deep pools (2.8-24 cm deep) increased species richness and abundance where they were present, highlighting the importance of water retaining features. Crevices and ledges were found to be significant in facilitating greater mobile species abundance. Integrating detailed microhabitat sampling into ecological surveys develops the understanding required to improve ecological enhancement opportunities alongside highlighting a greater need for these microhabitats to be better sampled in ecological surveys.

The conclusions of these findings enhance both ecological and biogeomorphological surveys, allowing a better understanding of how microhabitats influence species richness, abundance and overall distribution of species on natural rocky shore platforms, with these ecological variables found to be highly dependent on rock material and rock mass properties. This also enhances existing climate refugia studies as these results showcase the behavioural selection of microhabitats by intertidal species at low tide across multiple shores. Thermally sensitive intertidal species are expected to strongly respond to changes in climate as increased temperatures will result in increased body temperatures for organisms in the mid-upper intertidal zone where aerial exposure is prolonged, resulting in increased risks of thermal and desiccation stress (Harley and Helmuth, 2003; Helmuth et al., 2013; Helmuth and Hofmann, 2001). Although the behavioural responses of species to thermal stress and their resulting distribution on the shore is complex, with a high degree of small-scale variability (Helmuth et al., 2013), the results of Chapter 3 showcased the importance of deep pools, crevices and ledges as features. Pools and crevices offer the greatest refugia from stressful intertidal conditions, particularly where crevices hold water to maintain a damper microclimate. The consistency of these findings allows an improved understanding of small-scale responses to exposure at low tide across multiple shores.

From a geomorphological perspective, the jointing data from the Welsh limestone platform highlighted the significant congregation of limpets in discontinuities (e.g. associated crevices and shallow pools) when compared to the platform surface. Bioeroding species, such as limpets, have been noted to alter the morphology of pools and other features through grazing activity (Naylor et al., 2012b; Scheffers et al., 2012). However, the spatial variation in their distribution and density of organisms has limited quantification in geomorphological literature (Naylor et al., 2012b). The research in this thesis contributes to a better geomorphological understanding of the role of grazing organisms in weakening joints and the influence of biology in altering the morphology of shore platforms. Additionally, as crevices are shown to be important for these organisms on the limestone shore platform (which is softer and more chemically suitable for bioerosion), the species are gaining habitat and creating it through ecosystem engineering. This ultimately increases habitat suitability for the engineering species (Phillips, 2016). This work showcases that on softer lithologies, the concentration of bioerosive species in joints is indicative of a biogeomorphologically active zone. Although biological weathering was not measured, biology is most prevalent where geomorphic processes are most active (excluding abrasion zones) (Chapter 1.1), particularly with regards to bioerosive species in chemically suitable lithologies (Figure 5-27). In examining biogeomorphological feedbacks (Corenblit et al., 2011; Phillips, 2016), there is an apparent positive feedback between rock mass properties and microhabitats forming through geomorphic processes on softer lithologies (e.g. limestone), organisms occupying this space and engineering it, which would increase topographic complexity over time and provide new habitat for species to colonise.

On artificial shores, the results of Chapter 4 highlighted that artificial structures need to incorporate both the fine-scale (µm-mm) complexity that is important for early stage colonisation of barnacle species and the cm scale features that offer refuge and reduced desiccation stress that contribute to more diverse ecological communities (MacArthur et al., 2019). Chapter 4 of the thesis created eight designs (nine including clearings) of concrete and natural cement tiles of varying habitat complexity (low (mm) to high (cm's). The pits on high complexity tiles increased humidity and reduced the temperature compared to lower complexity designs and the tile surface (Figure 6-1). Species richness and abundance was greater in these microhabitats, which were frequented by gastropods across all sites from as early as 2 months from tile installation, with common shore crabs found at 6 months, highlighting the value of this habitat in attracting species to hard coastal structures that would otherwise be unable to support them. After 18 months, fucoid algae was found growing within pits at one site (Saltcoats), indicating that the higher complexity may promote faster succession than smoother designs. Intermediate complexity in the form of mm-scale grooves was the most effective design in increasing barnacle cover, with this significant difference notable from 2 months after installation. An optimised design to maximise ecological suitability would combine the mm-scale grooves with high complexity microhabitats at the cm-scale (Figure 6-1).

Chapter 5 introduced a novel approach to ecological enhancement of rock armour revetments via the passive positioning of boulders whose rock properties have been optimised for their habitat potential. Key findings show that boulder material choice should consider albedo, porosity and surface roughness as factors affecting long-term ecological engineering potential. Limestones are typically more ecologically suitable than granite, with Portland limestone and the Carboniferous limestones (Welsh and Hartlepool) both optimal boulder choices for combining ecological and engineering suitability from the laboratory results at the mm-cm (Figure 6-1). Adding in the field data on cm-dm scale features that was collected on several lithologies increased the ecological suitability ranking of several lithologies. In combining laboratory and field results, the Carboniferous limestone at Hartlepool was recommended but these results are more limited as only several lithologies were surveyed or observed in the field. Although Magnesian limestone had the highest ecological suitability overall from the laboratory and field tests (Chapter 5.6.1.), it was not suitable for engineering purposes, although would provide ecologically beneficial infill material between boulders.

A new sampling methodology was created to better quantify the influence of the positioning of the boulders (passive enhancement) at Hartlepool (Chapter 5) as quadrat sampling failed to capture the habitat preferences of species on boulders exceeding 2 m in width. Ledges at the scale of decimetres were key in significantly increasing limpet abundance, matching baseline values within 24 months, whereas the presence of other positioned features failed to influence species richness or abundance. As retrofitted grooves and pits/holes have been shown to increase species richness and abundance in previous studies (Firth et al., 2014b; Hall et al., 2018) due to the increased complexity of the surface, these features are recommended to be retrofitted in future. The presence of existing blast features should also be better positioned to retain water in order to improve their potential as habitat and function similar to retrofitted grooves (Hall et al., 2018). This combination of ecological sampling with improving the understanding of the underlying material properties and the role of topographic complexity, which are crucial in advancing ecological enhancement theory and habitat creation in practice.

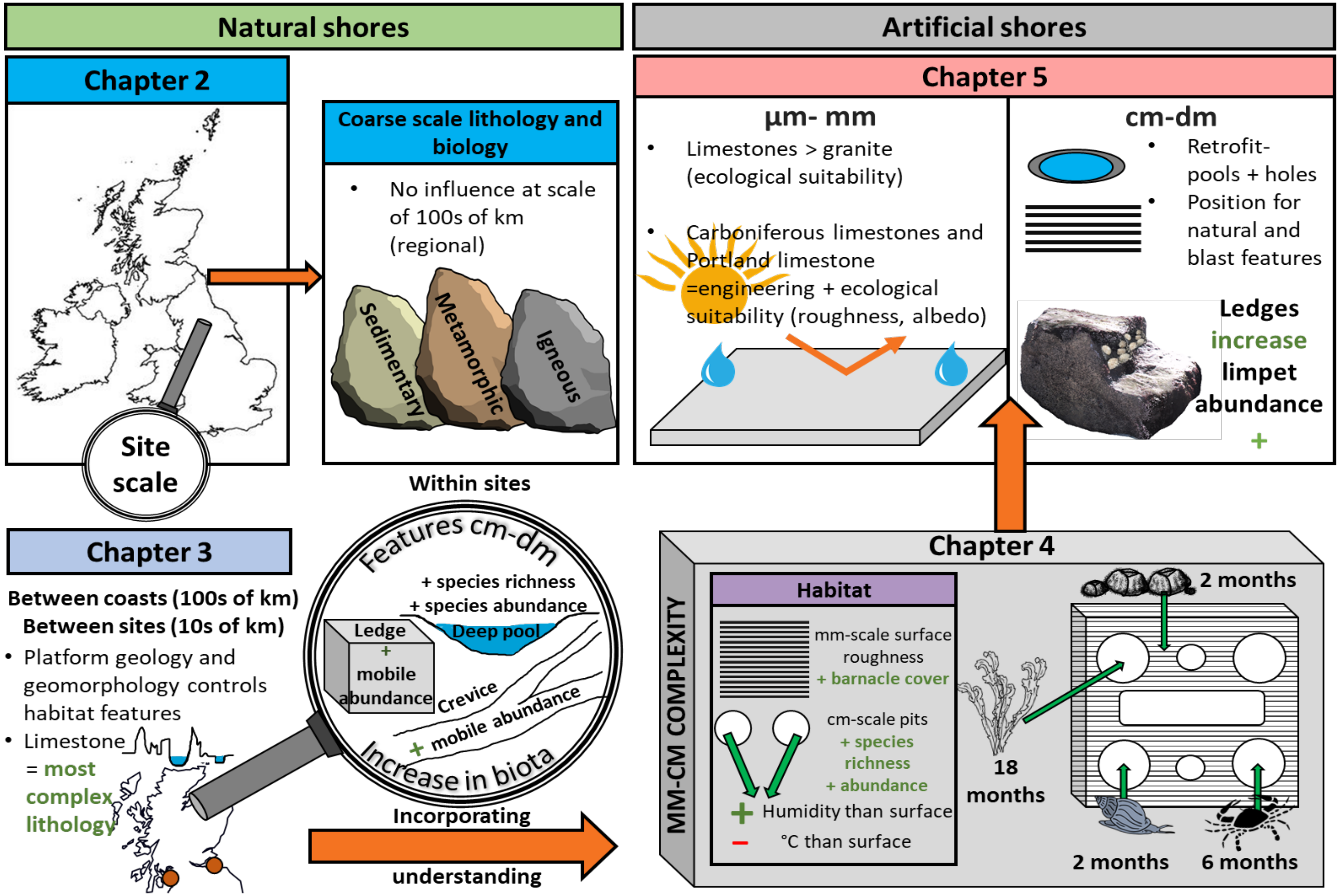

Figure 6-1. Flow of thesis chapters linking the understanding of interactions between lithology, habitat features and intertidal species on natural shores and the development of informed ecological enhancement on artificial shores.

6.2 Variability across spatial scales

Spatial scale has been an important constant in underpinning the theories and development of field strategies in this research. This research focused on the influence of rock material properties, rock mass properties and geomorphic features. This ranged from the influence of surface roughness at the µm-mm to lithology between shores (10s of kms). Although not examined in this research, the larger scale influences of climate, exposure and larval supply are important in influencing species richness and distributions (Figure 6-2).

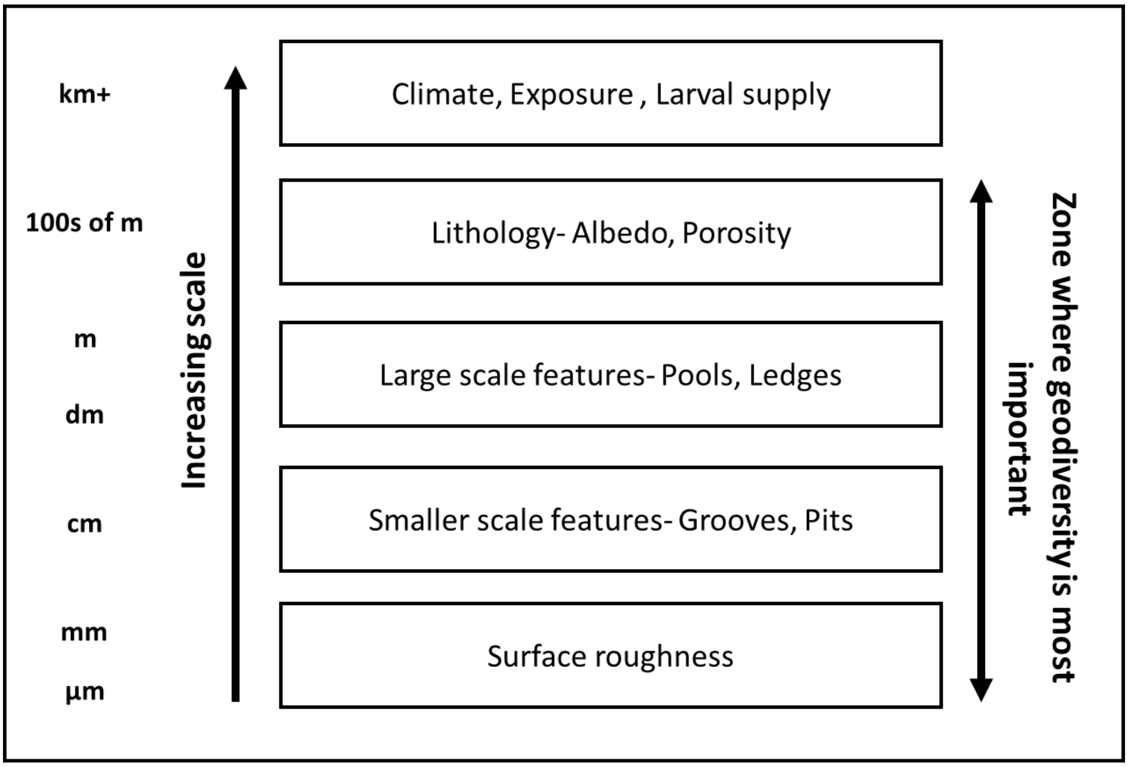

Figure 6-2. Physical and environmental factors that influence intertidal community composition with increasing spatial scale.

Organisms respond differently to their environments and individual species will have varying degrees of tolerance to exposure. For example, filter feeders typically do better in wave exposed conditions and large seaweeds are often more abundant in sheltered areas (Burrows et al., 2014). Wave exposure can alter the structure of communities (McQuaid and Branch, 1985), particularly in high wave energy environments where species on coastal defence structures are physically disturbed and dislodged (Hall et al., 2018). Larval supply is also highly variable and dependent on a range of factors including the physical transport of larvae by currents, distance to shore and the availability of suitable bare substratum (Jackson, 2015; Jonsson et al., 2004). Sheltered habitats may have reduced planktonic food supply and larval supply (Arribas et al., 2014) which often exerts a large effect over community development (Coombes et al., 2015).

The response of individual species is often climate related: two species exposed to the same climate may experience different levels of stress (Broitman et al., 2008) based on their physical tolerances to thermal and desiccation stress. Long-term changes in climate may cause range shifts in species distribution and abundance, with higher temperatures increasing the rate of mortalities of thermally sensitive species (Harley, 2008). This is particularly prevalent on darker lithologies which develop higher temperatures, particularly higher in the intertidal zone, thus reducing the abundance of species sensitive to thermal stress (Kordas et al., 2014). Lighter lithologies have lower surface temperatures that reduce the thermal stress experienced by species during low tide (Coombes et al., 2017; McGreevy, 1985). Higher porosity lithologies also reduce thermal stress by keeping the rock wetter for longer during low tide (Coombes and Naylor, 2012).

Settlement success is often influenced by substrate heterogeneity (Jackson, 2015), a factor that varies with lithology. Fine-scale surface roughness influences the settlement of early colonising species and rates of recruitment (Cacabelos et al., 2016; Coombes et al., 2015). Small scale differences at the cm-m scale are likely the result of behavioural responses to microhabitats which offer refuge from increased temperature during low tide, wave exposure and predation (Underwood and Chapman, 1996). The presence of refugia on natural and artificial shores including pits, crevices and pools influences species distribution and community structure, often resulting in increased biodiversity compared to adjacent areas of lower complexity (Chapman and Blockley, 2009; Firth et al., 2014b; Martins et al., 2007). However, the area-independent increases in biodiversity seen in relation to habitat complexity can vary between rocky shores (Kostylev et al., 2005), indicating that the effects of enhancement are highly variable in space.

6.3 Natural shore surveys

Different lithologies were found to have variation in the presence of different microhabitats across mm-dm+ scales (Figure 6-3). This variation is important since variability with features exerts an influence over the intertidal biota. The limestone shore at Girvan was found to be the most complex as this shore had a high number of ledges and pools that exceeded the dm scale (Table 6-1). Although in surveys there was no mm-scale hairline cracks, this shore had high mm-cm scale complexity within ledges (personal observation) that was not able to be captured adequately with quadrat sampling, despite the detailed nature of this sampling (Figure 3-3B). The Magnesian limestone shore at Hartlepool was surveyed during baseline sampling but not for microhabitat presence as this detailed survey method was developed later. On subsequent visits, the Magnesian limestone platform was abraded due to the ongoing construction of the scheme and so microhabitats could not be quantified. Consequently, the presence of microhabitats was estimated from photographs of the site. Magnesian limestone was found to have a high number of pools across the cm-dm+ scales and had high abundance of cm-scale features due to the bioerosive nature of limpets creating home scars on the rock (personal observation). This bioerosive capacity increased the topographic complexity across the Magnesian limestone shore platform and highlights the potential that biogeomorphological interactions can have in creating additional habitat. This high level of complexity was not consistent across all the limestones examined, with the Blue Lias limestone having a high abundance of mm-scale hairline cracks (Table 6-1) that were insignificant in influencing species distribution as these features had little refugia benefits (Figure 6-3).

Additional variation occurred between the two sandstones. The Barns Ness sandstone had a moderately high abundance of crevices at the cm's-dm scale but lacked habitat complexity at higher (dm+) and lower (mm-cm) levels. In contrast, the sandstone at Cove Harbour lacked features larger than the cm-scale but was highly pitted, offering small wetted refugia compared to the comparatively smoother platform surface. Although pits ranged from 2.5-5 cm in width and 1.2-5 cm in depth they were included in cm-scale features as they are smaller in scale than crevices and shallow pools when examining across width and depth measurements. In comparing the two igneous shores, the basalt had a higher abundance of mm-scale hairline cracks and dm+ scale pools and ledges than the andesite shore.

Table 6-1. Abundance of feature types (microhabitats) in each lithology by approximate scale from n=30 quadrats (25 cm^2). (S) = sedimentary, (I) = igneous lithology.

	mm scale	cm scale		cm's-dm scale		dm+ scale	
Lithology	**Hairline crack**	**Crack**	**Pit**	**Crevice**	**Shallow pool**	**Deep pool**	**Ledge**
Basalt (I)	26	6	0	1	6	10	24
Andesite (I)	2	0	0	0	5	8	15
Limestone (Girvan) (S)	0	0	13	2	9	13	29
Sandstone (S)	0	0	52	3	7	5	9
Blue Lias limestone (more jointed) (S)	11	7	0	26	16	0	2
Barns Ness limestone (S)	4	3	0	1	9	4	11
Barns Ness sandstone (S)	0	1	0	20	0	3	0

The overall trend across all shores was that sedimentary lithologies offered a greater variety of habitat types across multiple scales (mm-dm) (Figure 6-3). Softer lithologies are chemically suited to encourage ecological engineering, such as the high abundance of boring *Polydora ciliata* in pools on the Girvan limestone (Chapter 3.7) and high abundance of limpet home scars on the Magnesian limestone (personal observation). This biogeomorphological engineering of the rocks has greater potential to increase habitat heterogeneity with time, increasing ecological suitability over time as the lithologies become more easily weathered by a range of processes and actively shaped by biota (Coombes, 2011). This includes the potential for bioerosion by micro – macro organisms, which would be limited on harder lithologies such as andesite and basalt.

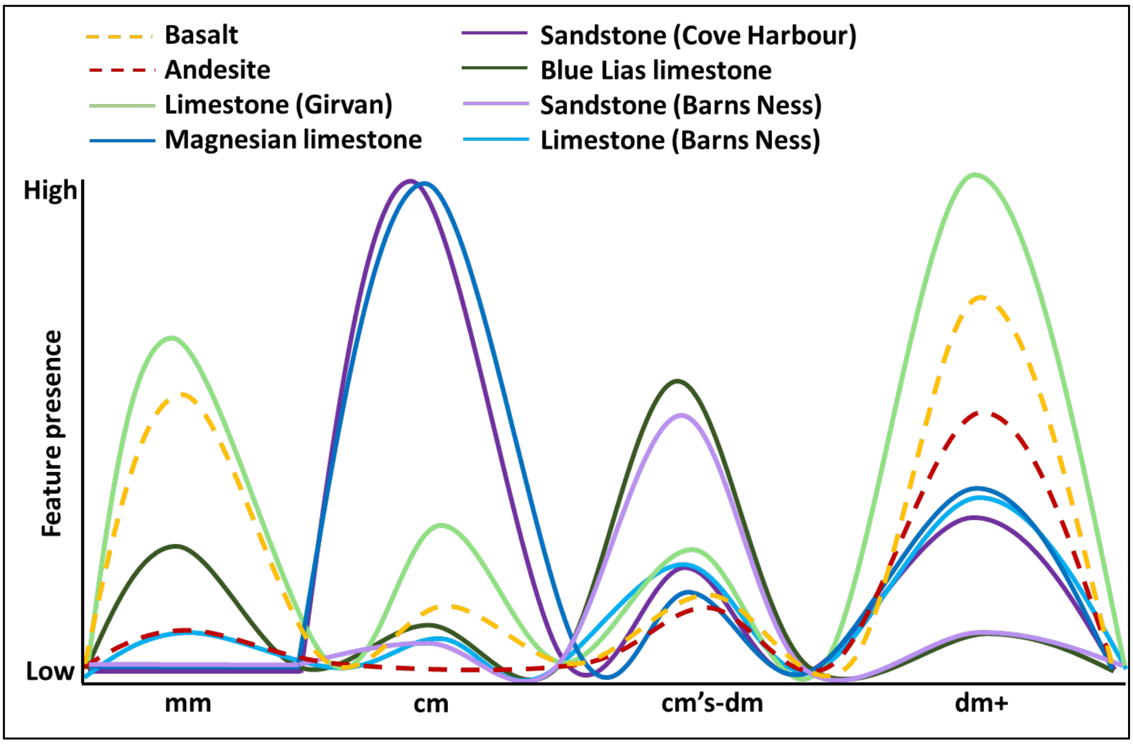

Figure 6-3. Schematic showing presence of geomorphic features from the mm-dm+ scale for natural shore lithologies examined in this thesis. Igneous lithologies are shown as dashed lines and sedimentary lithologies as solid lines.

6.4 Analysing the relative importance of habitat dimensions

This research has expanded the understanding of the behavioural selection of microhabitats by intertidal species, the key findings of which are presented in Chapter 3. However, to fully identify how natural features can better inform engineering design requires analysis into which dimensions (i.e. width, depth and water holding capacity of features) matter to rocky shore ecology in the mid-upper intertidal zone.

To assess this, the width and depth in mm and water holding capacity (measured as a percentage of the feature area within a 25 cm^2 quadrat) was recorded for every feature that fell within quadrats in Chapter 3. Features that performed well in attracting higher species richness or abundance, such as ledges, deep pools and crevices, were examined in greater depth (Figure 6-4). Pits were also examined due to their common use as retrofitted ecological enhancements on coastal infrastructure (Firth et al., 2014b; Hall et al., 2018; Loke et al., 2017; Martins et al., 2010). Generalised Linear Models (GLM) and Linear Models (LMs) were used where stated to examine the significance of width, depth and water holding percentage on species richness and abundance for each feature. Individual metrics were then grouped into depth or width bands (e.g. 100-200 mm) and tested for significance with further GLMs, LMs and Tukey's post-hoc test for multiple comparisons. These new analyses build on the findings from Chapter 3 to make informed recommendations on the optimum dimensions for different features in engineered design when drawing from natural rocky shore habitats.

A summation of the influence of the metrics measured on ecological variables is found in Figure 6-4. Although the width of features has limited influence on species distributions, it should still be considered in ensuring the dimensions of habitats fit the body size (Hacker and Steneck, 1990; Holling, 1992) and future growth of potential settling organisms. In addition, the refuge quality of individual habitats is variable and will be influenced by lithology and location but Figure 6-4 highlights that depth (of microhabitat) and water holding capacity are fundamental in optimising future engineering design.

Key results show that for pits, there was no specific factor of width, depth or water holding capacity that was more important for species richness (Table A 6-1). Water holding capacity (%) was noted to be more important in attracting increased mobile species abundance in pits than width or depth (Table A 6-1). For mobile species, the optimum width of pit habitat is between 30-40 mm in diameter and a depth of 10-20 mm (Table A 6-1). Statistically, 0-10% of water (n=12 pits) in the sampling area was better than 10-20% (n=2 pits) ($z=-3.930$, $p<0.001$) but patterns in the data show that species richness and mobile abundance is higher where more water is present (Figure A 6-1). Algae and lichen abundance was found in only two pits and so no statistical inference or patterns could be drawn from the association of algae and lichen with certain feature dimensions. Although retaining water is preferable, water does not have to fill the entire pit to create a wetter microclimate to benefit habitat (Chapter 4.11.5).

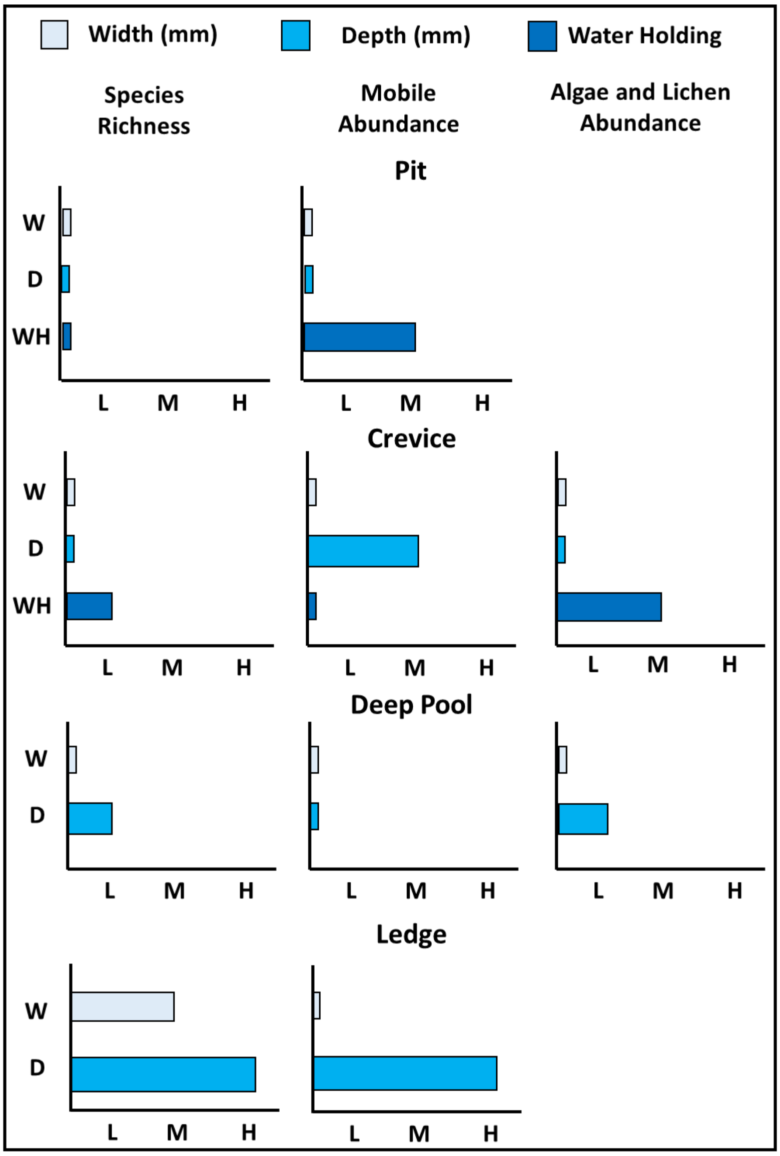

Figure 6-4. Influence of width, depth (proxy for height for ledges for consistency) and water holding capacity from low (L), moderate (M) to high (H) on species richness, mobile abundance and algae and lichen abundance for individual examined features.

Deep pools ranged from 70-1000 mm in width and 28-240 mm in depth (Table 3-1). In examining patterns within the data, 300-400 mm width had the highest species richness and abundance on average (Figure A 6-2). Depth was more variable and a minimum depth of 90 mm was determined to be optimal in promoting species richness, mobile abundance and algae and lichen abundance (Figure A 6-2A). This lack of significance with pool depth matches findings in drill-cored rock pools on artificial structures which found no significant difference in richness or community structure between artificial pools of 120 mm and 50 mm, although artificial pools supported higher species richness than adjacent artificial surfaces (Evans et al., 2015). Deeper pools are still recommended as they are more stable environments, reducing fluctuations in physical conditions including temperature, pH and salinity (Firth et al., 2014a). With this in mind, the optimum depth of engineered pools was recommended at 90 mm, with a minimum depth of 50 mm suggested. This minimum depth performed well in terms of average mobile abundance (Figure A 6-2A) and would be easier to implement into engineered designs (Evans et al., 2015).

Species richness and algae and lichen abundance were found to be more influenced by the water holding capacity of crevices than width or depth, although no specific proportion of water was significantly influential (Figure A 6-3). Mobile species abundance was more affected by the depth of crevices. Crevices with depths of 90-100 mm had greater species richness and mobile abundance than depths of 50-60 mm and 30-40 mm respectively (Table A 6-3, Figure A 6-3). Crevice depths of 90-100 mm were found to have greater algae and lichen abundance than all depths between 20 and 80 mm. In drawing patterns from the data, width and water holding capacity are unable to be divided into optimum measurements as they had variable influences on all ecological variables (Figure A 6-3). A minimum recommendation of 25 mm width is suggested as this is the minimum definition for a crevice in this research. Previous work including "grooves" on rock armour revetments that were sized at 10 mm deep and between 3 mm and 20 mm wide found grooves significantly increased species richness and abundance compared to control tests (non-manipulated boulder surface) (Hall et al., 2018). This scale of "groove" matches crack dimensions (Table 3-1), which were found to have limited influence on species richness and abundance (Chapter 3). However, the findings by Hall et al. (2018) highlight that lower depths will also produce significant differences in species richness and abundance. As the recommended depths of 90-100 mm may be difficult to implement across coastal defence schemes, it is suggested that schemes incorporate a variety of depths of crevice habitat, with a minimum of 30-40 mm to increase species richness and abundance and offer a wetter microclimate, as found in 30 mm pits in Chapter 4.

Species richness and mobile abundance were both influenced by the width and height of ledges (Table A 6-4). A width of 500-600 mm was found to have lower species richness than the smaller 100-200 mm wide ledges. However, ledge height was found to increase species richness compared to smaller ledges up to 100 mm. Mobile abundance was only influenced by height, with lower mobile abundance found on ledges under 100 mm tall (Table A 6-4, Figure A 6-4). It is likely that the trend of increasing mobile abundance with ledge height is related to increased surface area and overall habitat availability. Algae and lichen abundance was higher where ledge heights were between 500 and 600 mm (Figure A 6-4). Although widths exceeding 600 mm were moderately significant in increasing algae and lichen abundance, this was highly variable for species richness and mobile abundance (Figure A 6-4). Species distribution would also vary along the length of ledges and quadrat sampling is limited as it can only capture so much of species distribution. Overall, height is found to be a more important parameter for richness, mobile abundance and algae and lichen abundance and recommendations are based off this.

6.5 Rocky shore sampling methodology

Many ecological surveys of rocky shores, particularly those comparing natural and artificial habitats, examine biodiversity and species distributions within pools, crevices, pits and other microhabitats (Bergeron and Bourget, 1986; Davidson and Grupe, 2015; Evans et al., 2015; Firth et al., 2014a; Ostalé-Valriberas et al., 2018). However, many surveys also position sampling units away from features (Bulleri et al., 2005; Menge et al., 2010; Pister, 2009), standardising quadrats to avoid features like crevices and pools (Firth et al., 2016b; Scrosati et al., 2011). In other cases, such as the broad scale biological surveys in Chapter 2 (Burrows et al*.,* 2017), the specific features within which species were found were not recorded, instead an overview of what was present at each site was presented. These timed broad scale surveys are important in understanding species responses to larger scale phenomena including climate and exposure, but for future studies interested in the role of microhabitats in influencing intertidal biodiversity a hybrid sampling methodology would be optimal to a) better quantify geodiversity-biodiversity links on natural shores and b) to inform the design of engineered coastal and marine structures.

In examining ecological literature in the systematic review in Chapter 1 (Section 1.2), few studies note the underlying lithology of their study sites; yet, lithology has been demonstrated here to be an important determinant in microhabitat type and availability. Noting the features within which, or on which, species occur in rocky shore surveys is important as these habitats have been shown through this research to support a variety of species, offer refuge from intertidal stressors and for the most biologically suited lithologies, provide within-rock habitat for boring species. Ecological surveys could thus take better account of geological and geomorphological influences on species by noting both lithology and any surface or subsurface features. Broad scale surveys would also benefit from noting the microhabitats within which species are found in or on as a minimum requirement. More detailed surveys that are interested in the habitat requirements of species, biogeomorphic or geodiversity-biodiversity interactions, should additionally note the width, depth and water holding capacity as depth and water holding capacity were quantitatively shown to be of importance in influencing species richness and abundance (Section 6.4). The hardness and chemistry of the rocks is also recommended as a measure to identify lithologies which are most likely to have active biogeomorphic ecosystem engineers alongside identification of meso to macro-boring species.

This research clearly shows that cm-dm scale features such as pits, crevices, deep pools and ledge habitats offer the greatest habitat provision on semi-horizontal natural shores, with higher species richness and abundance than the adjacent platform surface and smaller-scale habitats (i.e. hairline cracks, cracks, ledge adjacent and pool adjacent habitats). These features should be noted where they are present and incorporated into more studies as they would enhance ecological and geomorphological understanding of biodiversity-geodiversity interactions (Section 6.1). Although cm-dm scale features are the optimal habitats to replicate on engineered structures, many of these larger features, such as ledges or deep pools, would not be possible to integrate into the design of engineered structures without retrofitting. Many locations, where wave exposure is high, would be unsuitable for retrofitting enhancements as this would be unsuitable on engineering grounds (damage and loss). Therefore, in integrating natural shore understanding into the enhanced design of artificial structure, design recommendations (Section 6.4) are context specific.

6.6 Ecological enhancement

Active ecological enhancement techniques via retrofitting of test tiles of different textures to inform future formliner manufacture (Chapter 4) and passive ecological enhancement through material selection and passive positioning of boulders (Chapter 5) were examined in this thesis. Tiles were tested on vertical seawalls where the surface complexity tested varied at the mm-cm scale across 8 designs. Here, barnacle abundance was found to be greatest on grooved designs of intermediate complexity (Figure 6-5). This is in line with previous studies where barnacles were positively correlated with roughness (Bell et al., 2015; Herbert and Hawkins, 2006), being significantly higher on mm-scale textures than smoother (Coombes et al., 2015) or more complex designs (MacArthur et al., 2019).

The cm-scale habitats up to 30 mm deep showed statistically significant increased species richness and abundance compared to smoother designs after 2 months. These match findings from initial trials in Singapore which found higher habitat complexity can support higher species richness, independent of surface area (Loke and Todd, 2016). These habitats were found to have a more suitable microclimate, i.e. lower temperature and higher humidity, which reduces the risk of desiccation stress and buffers the risk of predation and other environmental stressors (Cartwright and Williams, 2012; Kostylev et al., 2005; MacArthur et al., 2019). In effect, this represents the creation of refugia on typically plain-cast vertical seawalls and has likely contributed to the success in increasing species richness and abundance. Although the addition of finer-scale texture (mm-scale) and pits (up to 30 mm deep) may not have the same habitat value as larger scale features such as pools, they are still relevant in significantly increasing both the early settlement of barnacles and species richness and abundance onto artificial structures. These finer-scale (mm-cm scale) enhancements would be best suited for use on vertical structures and where wave exposure is too great for other features (e.g. bolt-on rock pools) to be added to structures (Figure 6-5). Where it is suitable oceanographically, increasing the scale (cm-dm+) and water holding capacity of vertical structures would increase biodiversity by increasing the diversity of microhabitats and surface area available for colonisation and settlement (Figure 6-5). Previous examples of this include Vertipools, a type of artificial rock pool which increased species richness compared to the adjacent seawall (Hall, 2017) and retrofitted flowerpots, which saw a

110% increase in species numbers within months of deployment (Browne and Chapman, 2011).

Ledge habitats increased mobile species abundance where passive positioning was implemented, likely as these features offered increased surface area for species to attach (dm+ scale) or on some shores offered shelter and a different orientation to waves. Other features on the examined rock armour such as blast features at Skinningrove (Section 6.1, Figure 6-1) and shallow pools (where water was retained in depressions up to approximately 1.5 cm deep) at Hartlepool made no influence on species richness or abundance. However, these features were not deliberately positioned (blast features) or scaled (pools) to create additional habitat value. Where holes and grooves, similar to blast features, have been deliberately retrofitted into previous studies, greater species richness and diversity occurred in these microhabitats compared to control surfaces (Hall et al., 2018), with pits and grooves increasing species abundance where they are present (Chapman and Blockley, 2009; Firth et al., 2014b; Hall et al., 2018).

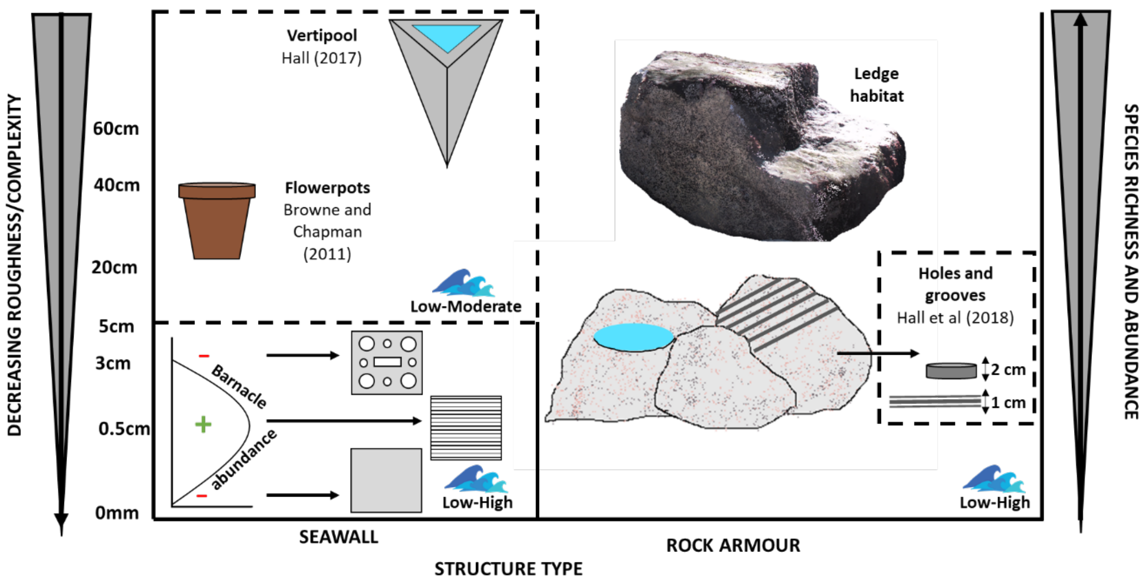

Figure 6-5. Schematic of the influence of roughness and complexity on species richness and abundance on the seawall and rock armour examined in this research. Dashed lines are representative of other studies that have shown the value of retrofitting habitat on these structures. Wave climate (exposure) in which features can be added are also indicated.

Retrofitting habitat at a higher scale of complexity is best implemented via the addition of water retaining features where wave exposure allows these features to be added without impacting on engineering function or durability (for example, a test array of 4 x 150 mm^3 blocks bolted onto the wall at Hartlepool were removed in one storm event). For example, pools are typically absent on most artificial structure types (especially walls and rock armour) yet the water these features retains increases their biodiversity potential (Firth et al., 2013; Hall, 2017; Loke et al., 2019). Artificial features should be designed and oriented so that they retain water, either to create a wetter microclimate (Chapter 4.11.5) or to match the species richness and abundance of natural pools, even though community composition will likely still differ (Evans et al., 2015). Varying the complexity, depth and size of these features will increase ecological niche availability (Bugnot et al., 2018) and extend the upper vertical limits of some species (Ostalé-Valriberas et al., 2018). Adding water retaining features may increase the presence of desiccation sensitive species and algal turfs; both of which are typically absent where these types of microhabitats are absent (Firth et al., 2016b). With increased water holding features, the potential to increase algal turf presence also promotes a more complex biogenic habitat, further enhancing refuge provision and feeding surfaces for macrofauna (Buschmann, 1990; Martinez, 2011). The presence of seaweeds has also been found to reduce weathering related risks to assets in the field (Coombes et al., 2013b), and to reduce the decay of rocks in laboratory trials (Gowell et al., 2015). Thus, the creation of artificial microhabitats suitable for these species groups has the ability to promote multi-functional ecological and engineering benefits.

6.7 Engineering recommendations

Based on the findings presented here, several recommendations can be made in relation to the specific dimensions of habitat features that can be added to vertical structures or rock armour revetments, emulating the width and depth dimensions of similar habitats on natural shores (Figure 6-6). Although 10-20 mm pit depth was found to be optimal for mobile species abundance (Section 6.4), the results from Chapter 4 indicated that 30 mm deep pits increased humidity and reduced temperature compared to the tile surface. With this in mind, 30 mm was the recommended depth for pits (Figure 6-6). Crevice width was recommended at 25 mm as this is the minimum definition of a crevice in this research. Recommendations are split between optimal and of value, showcasing where results may not have been significant but patterns in the data suggest that these dimensions will offer increased habitat compared to 'do nothing' approaches (Figure 6-6). These are suggested minimum requirements and acknowledge that the dimensions of habitat features forms only part of the ecological suitability "equation". The type of artificial habitat being constructed and the wave exposure of the shore (sheltered, moderate, high) should be considered, with species baseline surveys conducted at each site so that enhancements may benefit the species present. Retrofitting habitat using bolt-on designs (e.g. rock pools) are not suitable for high wave exposure conditions (Figure 6-6). Other features are suitable across a range of exposures (Figure 6-6) and can be designed for both vertical and artificial structures. Pits can be designed into formliners or retrofitted on rock armour. Crevice habitat can be created into formliners, likely limited to a depth of 30 mm on vertical structures or retrofitting 'grooves' into rock armour (Hall et al., 2018). Ledge-like habitat can be added to the base of seawalls as a concrete step and on rock armour, existing natural ledges on quarried boulders can be measured and positioned to enhance ecological suitability. For rock armour, lithology and its associated properties are also important factors, optimising for fine-scale texture, lighter colours and higher porosity where possible.

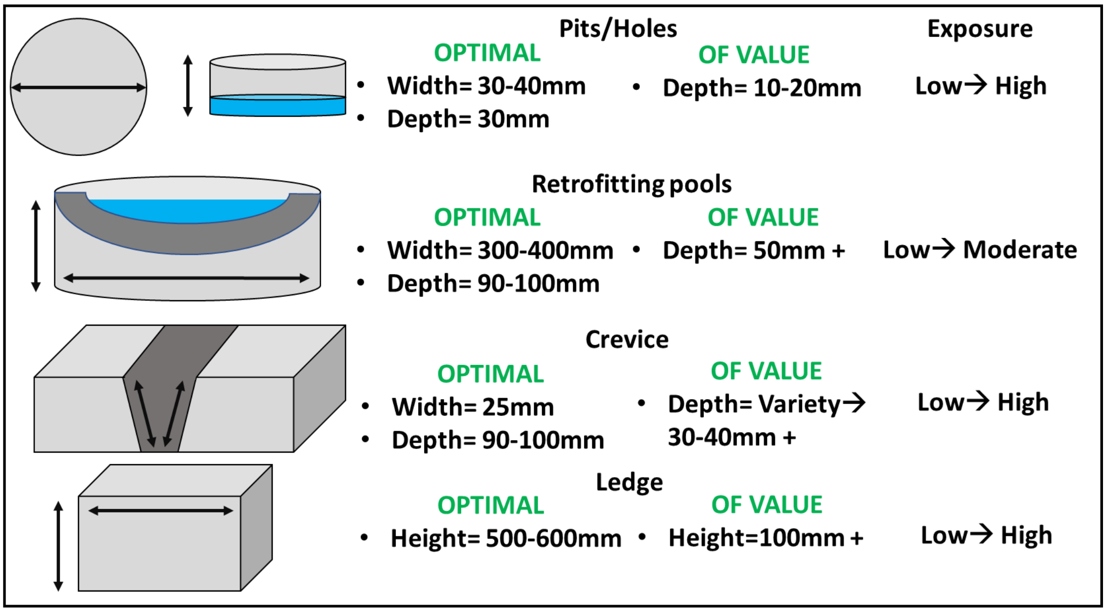

Figure 6-6. Recommendations of feature width and depth dimensions in mm alongside wave exposures they would be suitable in for recreating features on artificial structures that are comparable to natural shore microhabitats species richness and abundance. Optimal results based on findings from this research are presented as well as "of value" results, for where there are depth limitations in engineering designs.

Laboratory test results found that Magnesian limestone was optimal for ecology due to its high porosity and albedo. Magnesian limestone was also more chemically suitable and softer, adding to its potential for long term increases in surface roughness as softer lithologies have increased weathering and erodibility potential (Coombes, 2011), with visual evidence of active bioerosion creating surface roughness at the cm-scale (Figure 5-27), making it optimal for enhancing intertidal ecology. However, the potential for boring microorganisms and grazing species to increase the roughness of the rock serves to weaken the rock over time (Figure 5-27; Coombes et al., 2011). This lithology is also soft and erodible from a suite of geomorphic processes acting on it, with bed thickness too fine to produce boulders of large enough dimensions for use in a wave exposed environment, reducing the suitability of Magnesian limestone for engineering purposes. Of the remaining limestones tested, Portland limestone that was used in previous studies (Coombes et al., 2011; Coombes and Naylor, 2012) and is in widespread use in coastal engineering (e.g. Portland Port's Breakwater, and Plymouth Breakwater) and the Carboniferous limestones at Hartlepool and Wales served to optimise a combined ecological and engineering suitability. These lithologies had an ecologically suitable base roughness at the 25 mm^2 scale and suitably reflective albedos to reduce desiccation stress and are all suitable for blasting. However, Portland limestone has greater short (e.g. cyanobacteria boring had created mm-scale surface textures in as little as 20 months post-colonisation) and long term bioerosive potential, with its higher porosity and water absorption capacity making it more prone to weathering. This would increase the habitat complexity of the Portland limestone boulders, as Firth et al. (2013) stated any small depressions created form an important habitat for intertidal species. The long-term effects of this are readily apparent with high diversity and mobile abundance of species in limestone pools and more varied assemblages than concrete pools and adjacent substrate (Jackson, 2015). The Hartlepool and Welsh Carboniferous limestones had lower porosity and so may be more durable in the marine environment.

Enhancing rock armour involves the consideration of enhancement from the mm-dm+ scale by maximising surface roughness of the substrate and the existence of naturally occurring habitat features, and those from the blasting process. Smaller scale texture increases the suitability of materials for early stage colonisers (Coombes et al., 2015). Larger scale features increase the presence of key prey species, limpets in this study, for water birds. Passive positioning was beneficial in increasing the abundance of limpets, matching baseline values within 18 months although there was no influence on species richness. Species richness has been found to increase on structures where water retention is greater (Section 6.6) and so better positioning of blast features, limiting the free draining of blastholes and retrofitting pools would optimise the potential to maximise species richness.

With these factors in mind, a conceptual diagram of an optimised artificial shoreline is presented in Figure 6-7. Although the influence of specific ecological enhancement techniques will vary with climate, exposure and larval supply, this schematic presents a design that integrates the results of this research together with other studies (Coombes et al., 2015; Evans et al., 2015; Firth et al., 2014b; Hall et al., 2018; Loke and Todd, 2016). This includes the mm-cm scale texture that optimises early species colonisation and species richness and abundance on concrete formwork. In this research and previous work, this enhancement significantly increased macroinvertebrate abundance, creating more diverse communities than those found on smooth surfaces (Firth et al., 2014b; Moschella et al., 2005). Internal features angled downwards to retain water with a minimum height of 25 mm will allow organisms with this body size (Gastropods, limpets, anemones, mussels, crabs) to utilise this habitat. Rock armour boulders have been selected based on their baseline ecological suitability from rock material properties, with multiple lithologies incorporated into the design to combine ecological and engineering suitability. Blast features would be optimised by passively positioning to better retain water. Retrofitting pools, holes or grooves would also maximise species biodiversity potential. On other habitats such as concrete aprons in front of seawalls, retrofitted pits and grooves could also be created for the same purpose, adding a dip to the bottom of the concrete apron would function similarly to a crevice, providing a moist, shaded habitat (Figure 6-7).

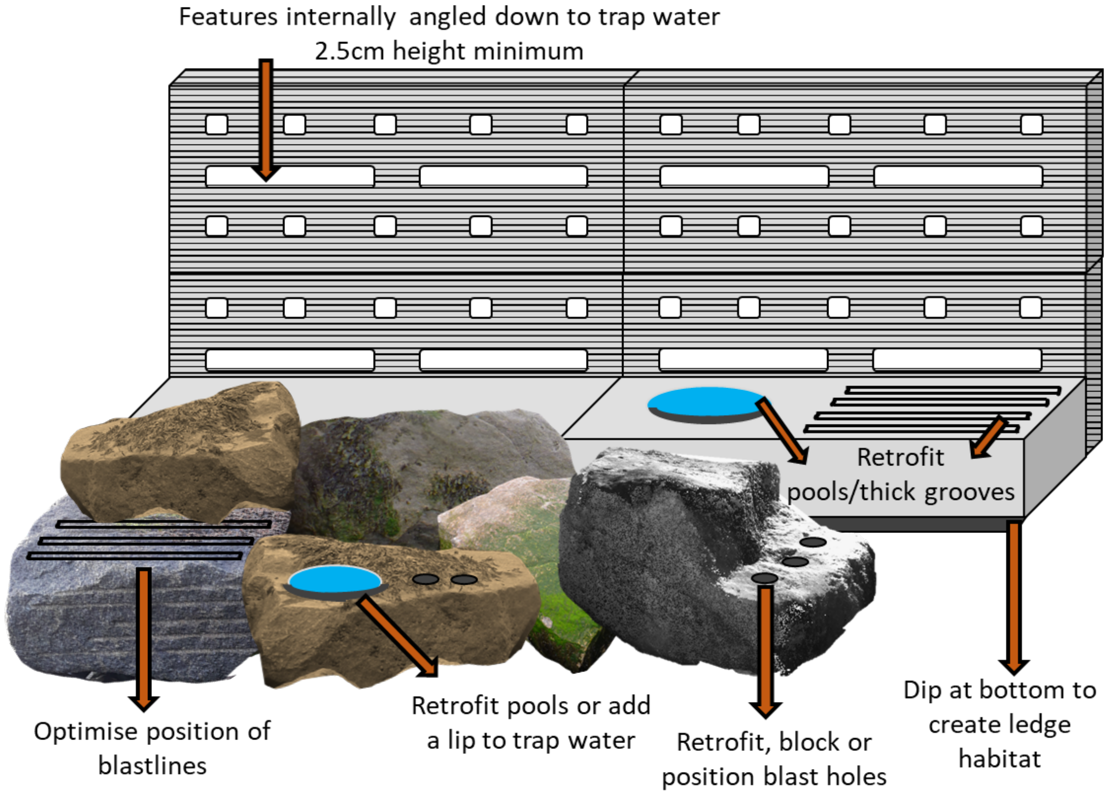

Figure 6-7. Conceptual diagram of an optimised artificial shoreline where seawalls and rock armour are enhanced for biodiversity, with multiple limestone lithologies on the revetment, mm-cm scale textured formliner and cm-dm scale enhancements on the concrete apron and rock armour boulders.

With enhancements based on the results from natural shores and their features, it is expected that the species in Table 6-2 would be expected to be capable of colonising most seawalls and rock armour revetments in a variety of locations. These results are based off the findings in Chapter 3, 4 and 5 and indicate expected artificial shore colonisers. Those included are relatively ubiquitous and a greater diversity of species will colonise each structure depending on the location of the site and type of enhancement used, with deeper pools and crevices likely to host a greater variety of species than smaller cm-scale pits. The timescale of expected colonisation assumes construction timings to commence at the beginning of annual settlement cycles for specific organisms. Water holding capacity is further highlighted as important in increasing species richness and abundance (Chapter 3.7, 3.9, 6.4). These habitats are effective in attracting common species including limpets and gastropods, albeit the exact species present will vary with location.

Table 6-2. Observed natural shore colonising species and expected colonisers of artificial shores (seawalls and rock armour revetments) based on the results from ecological enhancement studies and natural shore findings in this thesis and expected time scales (within months in brackets).

Feature	Observed natural shore colonisers		Expected artificial shore colonisers			
			Seawall		Rock armour	
	Mobile	**Sessile fauna and flora**	**Mobile**	**Sessile fauna and flora**	**Mobile**	**Sessile fauna and flora**
Pit	*Patella vulgata* *Littorina littorea* *Anurida maritima* *Gammarus* sp.	*Fucus spiralis* *Ulva* sp. Barnacles spp. *Actinia equina*	*Patella vulgata* **(2)** *Littorina littorea* **(2)** *Littorina saxatilis* **(2)** *Littorina obtusata* **(6)** *Gibbula umbilicalis* **(2)** *Melarhaphe neritoides* **(12)** *Nucella lapillus* **(18)** *Carcinus maenas* **(2)**	*Fucus spp.* **(18)** *Ulva sp.* **(2)** Barnacles spp. **(2)**	*Patella vulgata* **(2)** *Littorina littorea* **(2)** *Ligia oceanica* **(2)** *Talitrus saltator* **(2)** *Melarhaphe neritoides* **(6)** *Mytilus edulis* **(6)**	*Ulva sp.* **(2)** Barnacles sp. **(2)**
Crevice (Natural shore)/ Blastline (Artificial shore)	*Patella* spp. *Littorina littorea* *Littorina saxatilis* *Littorina obtusata* *Ligia oceanica* *Pomatoschistus microps* *Gibbula umbilicalis* *Nucella lapillus*	*Fucus spiralis* *Ulva* sp. *Pelvetia canaliculata* Barnacles spp. *Lichina pygmaea* *Rhodothamniella floridula* *Mytilus edulis* *Polydora ciliata* *Actinia equina* *Chondrus crispus* *Cladophora* sp.	*Patella vulgata* **(2)** *Littorina littorea* **(2)**	*Fucus spiralis* **(18)** *Ulva* sp. **(2)** Barnacles sp. **(2)** *Actinia equina* **(12)**	*Patella vulgata* **(2)** *Littorina littorea* **(2)** *Mytilus edulis* **(6)**	*Ulva* sp. **(2)** *Fucus spiralis* **(24)** *Fucus vesiculosus* **(24)** *Chondrus crispus* **(24)**

Shallow pool	*Patella* spp. *Littorina littorea* *Littorina saxatilis* *Littorina obtusata* *Gibbula umbilicalis* *Anurida maritima*	*Fucus spiralis* *Ulva* sp. *Pelvetia canaliculata* Barnacles spp. *Rhodothamniella floridula* *Polydora ciliata* *Actinia equina* *Chondrus crispus* *Corallina officinalis*	*Patella vulgata* **(2)** *Littorina littorea* **(2)**	*Fucus spiralis* **(24)** *Ulva* sp. **(2)** Barnacles sp. **(2)** *Actinia equina* **(12)**	*Patella vulgata* **(2)** *Littorina littorea* **(2)**	*Ulva* sp. **(2)** *Fucus spiralis* **(24)** *Fucus vesiculosus* **(24)** *Actinia equina*
Deep pool	*Patella vulgata* *Littorina littorea* *Littorina saxatilis* *Gibbula umbilicalis* *Carcinus maenas*	*Fucus spiralis* *Ulva* sp. *Pelvetia canaliculata* Barnacles spp. *Lichina pygmaea* *Rhodothamniella floridula* *Mytilus edulis* *Polydora ciliata* *Actinia equina* *Chondrus crispus* *Mastocarpus stellatus* *Cladophora* sp. *Corallina officinalis* Coralline crust	*Patella vulgata* **(2)** *Littorina littorea* **(2)**	*Fucus spiralis* **(24)** *Ulva* sp. **(2)** Barnacles sp. **(2)** *Actinia equina* **(12)**	*Patella vulgata* **(2)** *Littorina littorea* **(2)** *Ligia oceanica* **(2)** *Talitrus saltator* **(2)**	*Ulva* sp. **(2)** *Fucus spiralis* **(24)** *Fucus vesiculosus* **(24)**

Ledge	*Patella vulgata* *Littorina littorea* *Littorina saxatilis* *Ligia oceanica* *Gibbula umbilicalis* *Bdellidae* sp.	*Fucus spiralis* *Pelvetia canaliculata* Barnacles spp. *Mytilus edulis* *Actinia equina* *Chondrus crispus* *Mastocarpus stellatus*	*Patella vulgata* *Littorina littorea*	*Fucus spiralis* **(24)** *Ulva* sp. **(2)** Barnacles sp. **(2)**	*Patella vulgata* **(6)** *Littorina littorea* **(6)** *Littorina obtusata* **(6)**	*Fucus spiralis* **(24)** *Fucus vesiculosus* **(24)** *Ulva* sp. **(2)** Barnacles spp. **(2)** *Porphyra umbilicalis* **(6)**

6.8 Future work

Topographically complex intertidal habitats are technically difficult to sample due to heterogeneous nature of rocky shores. This research included the quantification of geomorphic features and species within them which are typically missed by standard ecological survey methods as quadrats are often positioned away from these features (Bulleri et al., 2005; Menge et al., 2010; Murray et al., 2001; Pister, 2009). However, the detailed methods of this thesis still underestimated the mm-cm complexity on the more complex limestone shore at Girvan as there were sub-features (high mm-scale complexity) within features. Further integration of Terrestrial Laser Scanning and more advanced methods of fine-scale habitat sampling could better quantify complexity and surface area on intertidal rocky shores: this research did not disentangle the effect of complexity from area. Terrestrial laser scanners could be better integrated with ecological surveys on rocky intertidal shores, which the work at Barns Ness offers a starting point for, as there are more opportunities for structurally explicit capturing of geomorphic complexity of coast and the importance of this for ecology. Detailed microclimate data would also improve the quality of this research and future studies by mapping temperature and humidity across the shore and in various microhabitats, which would allow an improved understanding of the refuge function of these geomorphic features.

Along with this, further lithological testing could be integrated: time constraints here meant the rocks on several shores were not tested for albedo, porosity and surface roughness that may have provided further insight into their ecological suitability. The influence of bioeroders appeared limited at the scales examined but sub-mm scale examination of microboring species would benefit understanding of the full range of spatial scales at which biogeomorphological processes are known to operate on rocky shores (e.g. Coombes et al., 2011; Moura et al., 2012; Naylor et al., 2012b). This would allow for better recommendations on the ecological suitability of different lithologies as softer lithologies have greater ecological engineering potential and consequently long-term ecological suitability.

Scaling up enhancement trials to full construction schemes requires longer term monitoring as existing research provides a spatially (patch-scale microhabitat additions, Strain et al., 2017b) and temporally limited evidence base. On engineered structures, longer-term monitoring would benefit the understanding of how enhancements operate ecologically across the design life of structures (approximately 50-100 years) and whether enhancements impact engineering structural integrity. Short-term ecological enhancement studies also limit the ability to determine long-term biodiversity benefits and see how successional communities develop and when communities stabilise (Hall, 2017; Strain et al., 2017b), highlighting the need for longer monitoring. There may be limits to how much seawalls can be (re-) textured and so retrofitting habitat needs better areal consideration if these designs are to be scaled up for commercial application.

The interdisciplinary nature of this research has shown to provide important findings for both geomorphology and ecology (Section 6.1). However, even within this research terminology differences (e.g. microhabitats/geomorphic features, topographic complexity/geomorphic complexity) are evident between ecologists and geomorphologists. Future research would benefit from further collaboration between ecologists, geomorphologists and engineers to combine research and knowledge to enhance blue skies research and the development of larger scale ecologically enhanced designs.

6.9 Conclusion

This thesis met the overarching aim of developing the understanding between biodiversity and geodiversity interactions in the mid-upper intertidal zone of natural and artificial shores. Strong relationships were found between the presence of microhabitats and species response in both natural and artificial habitats. On natural shores, deep pools, crevices and ledges had positive influences on species richness and abundance compared to the adjacent platform and smaller scale microhabitats.

The research in this thesis provides biogeomorphological insights into the role of bioerosive species in weakening joints. Limpets were highly concentrated in discontinuities, where geomorphic processes are most active, which on chemically suitable lithologies, such as the limestone platform at Glamorgan, is indicative of a biogemorphologically active zone over longer time periods. Additionally, lithology and its associated properties was important in generating geomorphic features, which in turn promote biota. This highlights the importance of including rock material and rock mass properties in ecological studies and the need for more detailed habitat sampling on natural shores. Ecological surveys would additionally benefit from noting the width, depth and water holding capacity of individual features as these exert varying influences on species and would allow for greater understanding of species survival and distributions on rocky shores.

Underlying lithological properties including albedo, porosity and surface roughness need better consideration in ecological and enhancement studies as these are important contributors to baseline ecological suitability. Where rock armour is required, lighter, rougher and more porous lithologies with natural or engineered surface features maximise surface roughness across multiple scales. To optimise for species richness and abundance on artificial structures, habitat complexity should be maximised by incorporating a diverse range of microhabitats from the cm-dm scale underpinned by mm-scale surface roughness. If hard coastal structures are required over more nature-based solutions (e.g. managed realignment), experimental trials from this research (Chapters 4 and 5) improve the scientific confidence of ecological enhancement strategies by showcasing their significance in increasing species richness and abundance compared to standard engineering designs. This allows for more ecologically and geomorphologically informed commercial scale applications of these enhancements in future.

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Chapter 1 Appendix

Table A 1-1. References used in systematic review in Chapter 1.

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Chapter 2 Appendix

Table A 2-1. Summary of the results of Quasi-Poisson and Negative Binomial GLMs for species richness and abundance metrics with lithology as a factor in the national dataset (2014/2015) (NS=Not significant).

a) Total species richness- Negative Binomial				
Comparison	**Estimate**	**Std. Error**	**Z value**	**P-value**
Metamorphic- Igneous	-0.031	0.044	-0.711	NS
Sedimentary- Igneous	-0.007	0.043	-0.168	NS
Sedimentary- Metamorphic	0.024	0.036	0.676	NS
b) Algae and lichen richness - Negative Binomial				
Comparison	**Estimate**	**Std. Error**	**Z value**	**P-value**
Metamorphic- Igneous	-0.023	0.055	-0.465	NS
Sedimentary- Igneous	-0.020	0.054	-0.357	NS
Sedimentary- Metamorphic	0.006	0.044	0.141	NS
c) Sessile fauna richness- Quasi-Poisson				
Comparison	**Estimate**	**Std. Error**	**Z value**	**P-value**
Metamorphic- Igneous	-0.025	0.076	-0.330	NS
Sedimentary- Igneous	-0.088	0.075	-1.168	NS
Sedimentary- Metamorphic	-0.063	0.063	-1.006	NS
d) Other (typically lower shore species)- Quasi-Poisson				
Comparison	**Estimate**	**Std. Error**	**Z value**	**P-value**
Metamorphic- Igneous	0.152	0.139	1.093	NS
Sedimentary- Igneous	0.052	0.139	0.372	NS
Sedimentary- Metamorphic	-0.100	0.109	-0.920	NS
e) Total abundance - Negative Binomial				
Comparison	**Estimate**	**Std. Error**	**Z value**	**P-value**
Metamorphic- Igneous	-0.053	0.051	-1.032	NS
Sedimentary- Igneous	0.003	0.050	0.063	NS
Sedimentary- Metamorphic	0.056	0.041	1.357	NS
f) Algae and lichen abundance- Negative Binomial				
Comparison	**Estimate**	**Std. Error**	**Z value**	**P-value**
Metamorphic- Igneous	-0.039	0.064	-0.609	NS
Sedimentary- Igneous	-0.014	0.063	-0.215	NS
Sedimentary- Metamorphic	0.025	0.051	0.492	NS
g) Sessile fauna abundance- Negative Binomial				
Comparison	**Estimate**	**Std. Error**	**Z value**	**P-value**
Metamorphic- Igneous	0.060	0.099	0.610	NS
Sedimentary- Igneous	-0.021	0.099	-0.214	NS
Sedimentary- Metamorphic	-0.082	0.080	-1.020	NS
h) Barnacle abundance- Negative Binomial				
Comparison	**Estimate**	**Std. Error**	**Z value**	**P-value**
Metamorphic- Igneous	-0.104	0.070	-1.479	NS
Sedimentary- Igneous	-0.060	0.069	-0.868	NS
Sedimentary- Metamorphic	0.044	0.058	0.770	NS
i) Other abundance- Negative Binomial				

Comparison	Estimate	Std. Error	Z value	P-value
Metamorphic- Igneous	0.082	0.172	0.475	NS
Sedimentary- Igneous	0.069	0.169	0.406	NS
Sedimentary- Metamorphic	-0.013	0.138	-0.094	NS

Table A 2-2. Summary of the results of Quasi-Poisson and Negative Binomial GLMs for species richness metrics with region (East Scotland, West Scotland and South Wales) and grouped lithology (igneous, not-igneous) as factors in the regional dataset (***=p<0.001,**=p<0.01, *=p<0.05, NS= Not significant).

a) Total species richness- Quasi-Poisson				
	Estimate	**Std. Error**	**T value**	**P-value**
(Intercept)	2.442	0.193	12.629	***
South Wales	0.841	0.197	4.276	***
West Scotland	0.123	0.225	0.545	NS
Not igneous	0.218	0.208	1.046	NS
South Wales: Not igneous	-0.210	0.213	-0.989	NS
West Scotland: Not igneous	-0.126	0.239	-0.526	NS
b) Mobile species richness- Quasi-Poisson				
	Estimate	**Std. Error**	**T value**	**P-value**
(Intercept)	0.916	0.221	4.145	***
South Wales	1.339	0.223	5.998	***
West Scotland	0.247	0.253	0.975	NS
Not igneous	0.215	0.238	0.903	NS
South Wales: Not igneous	-0.253	0.241	-1.048	NS
West Scotland: Not igneous	0.011	0.269	0.039	NS
c) Sessile fauna richness (Mussels, Anemones, Oysters) - Quasi-Poisson				
	Estimate	**Std. Error**	**T value**	**P-value**
(Intercept)	7.820 e-15	0.443	0.000	NS
South Wales	9.008 e-01	0.449	2.005	*
West Scotland	1.823 e-01	0.511	0.357	NS
Not igneous	9.531 e-02	0.481	0.198	NS
South Wales: Not igneous	7.613 e-02	0.490	0.156	NS
West Scotland: Not igneous	1.245 e-01	0.547	0.227	NS
d) Barnacle species richness - Quasi-Poisson				
	Estimate	**Std. Error**	**T value**	**P-value**
(Intercept)	-2.228 e-14	0.300	0.000	NS
South Wales	1.495	0.303	4.936	***
West Scotland	0.876	0.325	2.698	**
Not igneous	0.340	0.321	1.048	NS
South Wales: Not igneous	-0.331	0.325	-1.021	NS
West Scotland: Not igneous	-0.413	0.345	-1.199	NS
e) Algae and lichen richness- Quasi-Poisson				
	Estimate	**Std. Error**	**T value**	**P-value**
(Intercept)	1.946	0255	7.625	***
South Wales	0.273	0.263	1.038	NS

West Scotland	-0.121	0.308	-0.395	NS
Not igneous	0.134	0.277	0.483	NS
South Wales: Not igneous	-0.144	0.286	-0.504	NS
West Scotland: Not igneous	-0.112	0.328	-0.343	NS

Table A 2-3. Summary of the results of Quasi-Poisson and Negative Binomial GLMs for species abundance metrics with region (East Scotland, West Scotland and South Wales) and grouped lithology (igneous, not-igneous) as factors in the regional dataset (***=p<0.001, *=p<0.05, NS= Not significant).

a) Total abundance - Negative Binomial

	Estimate	**Std. Error**	**T value**	**P-value**
(Intercept)	3.784	0.179	21.205	***
South Wales	0.739	0.184	4.019	***
West Scotland	0.136	0.210	0.647	NS
Not igneous	0.271	0.194	1.396	NS
South Wales: Not igneous	-0.229	0.201	-1.138	NS
West Scotland: Not igneous	-0.157	0.225	-0.697	NS

b) Mobile abundance - Negative Binomial

	Estimate	**Std. Error**	**T value**	**P-value**
(Intercept)	2.303	0.238	9.660	***
South Wales	1.270	0.242	5.255	***
West Scotland	0.322	0.272	1.184	NS
Not igneous	0.239	0.257	0.930	NS
South Wales: Not igneous	-0.261	0.261	-0.996	NS
West Scotland: Not igneous	-0.124	0.290	-0.427	NS

c) Algae and lichen abundance- Negative Binomial

	Estimate	**Std. Error**	**T value**	**P-value**
(Intercept)	3.258	0.303	10.743	***
South Wales	0.206	0.314	0.657	NS
West Scotland	-0.140	0.361	-0.389	NS
Not igneous	0.253	0.331	0.766	NS
South Wales: Not igneous	-0.277	0.345	-0.803	NS
West Scotland: Not igneous	-0.114	0.386	-0.295	NS

d) Barnacle abundance- Quasi-Poisson

	Estimate	**Std. Error**	**T value**	**P-value**
(Intercept)	1.609	0.284	5.666	***
South Wales	1.240	0.287	4.316	***
West Scotland	0.713	0.311	2.295	*
Not igneous	0.199	0.306	0.649	NS
South Wales: Not igneous	-0.147	0.311	-0.474	NS
West Scotland: Not igneous	-0.279	0.333	-0.839	NS

e) Sessile fauna abundance (Anemones, Mussels, Oysters) - Negative Binomial

	Estimate	**Std. Error**	**T value**	**P-value**

(Intercept)	1.099	0.444	2.472	*
South Wales	0.592	0.455	1.301	NS
West Scotland	0.236	0.512	0.461	NS
Not igneous	-0.105	0.491	-0.215	NS
South Wales: Not igneous	0.626	0.503	1.244	NS
West Scotland: Not igneous	0.392	0.555	0.706	NS

Table A 2-4. SIMPER analysis on full ecological community between sedimentary and igneous lithologies on the east coast of Scotland. Species are listed in order of their contribution to dissimilarities between assemblages. Average= species contribution to between-group dissimilarity, sd= standard deviation of contribution, ratio= average:sd ratio, Sedimentary mean abundance, Igneous mean abundance, cum sum= ordered cumulative contribution and trophic level of species.

				East coast- Sedimentary- Igneous				
Species	**Average**	**sd**	**ratio**	**Sed mean abu**		**Ign mean abu**	**Cum sum**	**Trophic level**
Fuves	0.042	0.009	4.635	4.818	>	0.5	0.105	1
Pecan	0.030	0.021	1.431	1.636	<	4.5	0.179	1
Hasil	0.029	0.020	1.477	2.909	>	0.0	0.252	1
Nulap	0.026	0.015	1.723	2.818	>	0.0	0.316	3
Fuser	0.025	0.024	1.037	3.818	>	2.5	0.379	1
Hapan	0.024	0.017	1.361	2.545	>	0.0	0.437	2
Hielo	0.023	0.021	1.115	2.182	>	2.0	0.495	1
Myspp	0.022	0.020	1.093	1.818	<	2.0	0.548	2
Lilit	0.021	0.019	1.110	4.545	>	3.0	0.599	2
Ladig	0.021	0.023	0.890	3.091	<	4.5	0.650	1
Lisax	0.020	0.019	1.027	0.727	<	2.0	0.699	2
Chcri	0.019	0.018	1.040	1.909	>	1.5	0.746	1
Papal	0.015	0.013	1.114	1.273	>	1.0	0.782	1
Asnod	0.014	0.014	0.989	1.455	>	1.0	0.817	1
Maste	0.012	0.017	0.740	3.182	<	4.0	0.848	1
Acequ	0.012	0.011	1.168	1.455	>	1.0	0.878	3
Elmod	0.008	0.014	0.588	0.909	>	0.0	0.899	2
Fuspi	0.008	0.012	0.698	4.273	>	4.5	0.919	1
Alesc	0.006	0.013	0.451	0.636	>	0.0	0.934	1
Lipyg	0.005	0.011	0.460	0.545	>	0.0	0.945	1
Chmon	0.005	0.013	0.369	0.545	>	0.0	0.957	2
Padep	0.004	0.013	0.309	0.455	>	0.0	0.967	2
Pauly	0.004	0.013	0.309	0.455	>	0.0	0.977	2
Giumb	0.004	0.013	0.309	0.455	>	0.0	0.987	2
Gicin	0.002	0.004	0.459	0.182	>	0.0	0.991	2
Salat	0.002	0.006	0.308	0.182	>	0.0	0.995	1
Pavul	0.001	0.003	0.308	4.909	<	5.0	0.998	2
Chste	0.001	0.003	0.309	0.091	>	0.0	1.000	2
Lahyp	0.000	0.000	NA	0.000	=	0.0	1.000	1

Anvir	0.000	0.000	NA	0.000	=	0.0	1.000	2
Sebal	0.000	0.000	NA	5.000	=	5.0	1.000	2

Table A 2-5. SIMPER analysis on full ecological community between metamorphic and sedimentary lithologies on the west coast of Scotland. Species are listed in order of their contribution to dissimilarities between assemblages. Average= species contribution to between-group dissimilarity, sd= standard deviation of contribution, ratio= average:sd ratio, Sedimentary mean abundance, Igneous mean abundance, cum sum= ordered cumulative contribution and trophic level of species.

			West coast- Metamorphic- Sedimentary					
Species	**Average**	**sd**	**ratio**	**Met mean abu**		**Sed mean abu**	**Cum sum**	**Trophic level**
Ladig	0.024	0.022	1.063	2.125	<	2.370	0.064	1
Fuves	0.023	0.019	1.197	2.681	<	2.889	0.127	1
Fuser	0.023	0.022	1.027	3.639	<	4.037	0.190	1
Asnod	0.021	0.017	1.254	2.375	<	2.889	0.267	1
Myspp	0.021	0.016	1.278	2.306	<	3.037	0.304	2
Maste	0.020	0.016	1.260	2.333	<	2.556	0.359	1
Fuspi	0.019	0.018	1.089	3.611	<	4.370	0.411	1
Giumb	0.019	0.016	1.173	1.931	>	1.815	0.463	2
Acequ	0.019	0.015	1.280	2.403	<	2.482	0.514	3
Chmon	0.017	0.016	1.113	3.194	<	3.370	0.561	1
Pecan	0.015	0.017	0.872	4.375	>	4.333	0.601	1
Nulap	0.014	0.012	1.156	3.194	<	3.444	0.640	3
Lipyg	0.014	0.016	0.855	1.028	<	1.111	0.678	1
Chcri	0.013	0.014	0.951	0.736	<	1.370	0.714	1
Lisax	0.012	0.016	0.758	0.861	<	0.963	0.748	2
Lilit	0.011	0.012	0.942	4.042	<	4.815	0.779	2
Pavul	0.010	0.012	0.797	4.347	>	4.222	0.806	2
Hapan	0.010	0.014	0.712	0.417	<	1.000	0.832	2
Elmod	0.009	0.013	0.714	0.708	>	0.556	0.857	1
Gicin	0.008	0.012	0.701	0.389	<	0.778	0.880	2
Hasil	0.008	0.012	0.647	0.819	>	0.333	0.902	1
Alesc	0.008	0.014	0.560	0.639	>	0.444	0.923	1
Papal	0.005	0.011	0.519	0.306	<	0.482	0.938	1
Chste	0.005	0.012	0.450	0.486	>	0.259	0.952	2
Salat	0.004	0.009	0.414	0.306	>	0.185	0.963	1
Sebal	0.004	0.006	0.563	4.958	>	4.889	0.972	2
Mener	0.003	0.009	0.348	0.069	<	0.370	0.981	2
Hielo	0.003	0.009	0.306	1.194	>	0.148	0.989	1
Anvir	0.002	0.009	0.236	0.056	<	0.185	0.994	2
Pauly	0.002	0.007	0.307	0.125	<	0.148	1.000	2
Lahyp	0.000	0.000	NA	0.000	=	0.000	1.000	1
Padep	0.000	0.000	NA	0.000	=	0.000	1.000	2

Table A 2-6. SIMPER analysis on full ecological community between metamorphic and igneous lithologies on the west coast of Scotland. Species are listed in order of their contribution to dissimilarities between assemblages. Average= species contribution to between-group dissimilarity, sd= standard deviation of contribution, ratio= average:sd ratio, Sedimentary mean abundance, Igneous mean abundance, cum sum= ordered cumulative contribution and trophic level of species.

				West coast- metamorphic- igneous				
Species	**Average**	**sd**	**ratio**	**Met mean abu**		**Ign mean abu**	**Cum sum**	**Trophic level**
Ladig	0.029	0.024	1.194	2.125	<	3.4	0.072	1
Asnod	0.025	0.020	1.240	2.375	>	2.2	0.134	1
Fuser	0.024	0.023	1.084	3.639	<	3.8	0.194	1
Fuves	0.024	0.020	1.219	2.681	>	1.2	0.254	1
Fuspi	0.024	0.018	1.315	3.611	>	3.0	0.313	1
Giumb	0.023	0.019	1.224	1.931	<	2.6	0.370	2
Myspp	0.022	0.019	1.190	2.306	>	1.6	0.426	2
Maste	0.022	0.017	1.261	2.333	>	1.6	0.479	1
Chmon	0.022	0.017	1.262	3.194	>	2.8	0.532	2
Pecan	0.021	0.018	1.162	4.375	>	3.4	0.584	1
Nulap	0.020	0.017	1.206	3.194	<	3.8	0.633	3
Acequ	0.018	0.013	1.281	2.403	>	2.2	0.677	3
Lilit	0.016	0.018	0.916	4.042	>	3.4	0.717	2
Lipyg	0.015	0.014	1.069	1.023	<	1.4	0.755	1
Pavul	0.015	0.018	0.805	4.347	>	3.8	0.791	2
Hasil	0.014	0.014	0.973	0.819	<	1.2	0.825	1
Elmod	0.013	0.017	0.719	0.708	<	0.8	0.856	2
Alesc	0.010	0.013	0.804	0.639	<	0.8	0.882	1
Chste	0.010	0.017	0.608	0.486	<	0.8	0.908	2
Chcri	0.010	0.014	0.716	0.736	>	0.6	0.932	1
Lisax	0.008	0.015	0.524	0.861	>	0.0	0.951	2
Hapan	0.004	0.009	0.417	0.417	>	0.0	0.960	2
Gicin	0.004	0.009	0.376	0.389	>	0.0	0.969	2
Pauly	0.003	0.007	0.424	0.125	<	0.2	0.977	2
Salat	0.003	0.007	0.391	0.306	>	0.0	0.983	1
Papal	0.002	0.008	0.312	0.306	>	0.0	0.990	1
Sebal	0.002	0.004	0.434	4.958	<	5.0	0.994	2
Hielo	0.001	0.006	0.245	0.194	>	0.0	0.997	1
Mener	0.001	0.005	0.118	0.069	>	0.0	0.999	2
Anvir	0.001	0.003	0.168	0.056	>	0.0	1.000	2
Lahyp	0.000	0.000	NA	0.000	=	0.0	1.000	1
Padep	0.000	0.000	NA	0.000	=	0.0	1.000	2

Table A 2-7. SIMPER analysis on full ecological community between sedimentary and igneous lithologies on the west coast of Scotland. Species are listed in order of their contribution to dissimilarities between assemblages. Average= species contribution to between-group dissimilarity, sd= standard deviation of contribution, ratio= average:sd ratio, Sedimentary mean abundance, Igneous mean abundance, cum sum= ordered cumulative contribution and trophic level of species.

West coast- sedimentary-igneous								
Species	**Average**	**sd**	**ratio**	**Sed mean abu**		**Ign mean abu**	**Cum sum**	**Trophic level**
Ladig	0.028	0.023	1.173	2.370	<	3.400	0.067	1
Fuspi	0.025	0.018	1.355	4.370	>	3.000	0.128	1
Asnod	0.025	0.019	1.312	2.889	>	2.200	0.187	1
Fuves	0.024	0.019	1.290	2.889	>	1.200	0.246	1
Myspp	0.024	0.018	1.337	3.037	>	1.600	0.305	2
Fuser	0.022	0.022	1.040	4.037	>	3.800	0.359	1
Maste	0.022	0.018	1.197	2.556	>	1.600	0.413	1
Pecan	0.022	0.018	1.216	4.333	>	3.400	0.466	1
Giumb	0.022	0.018	1.189	1.815	<	2.600	0.519	2
Chmon	0.020	0.016	1.241	3.370	>	2.800	0.568	2
Nulap	0.018	0.016	1.109	3.444	<	3.800	0.611	3
Acequ	0.018	0.014	1.254	2.482	>	2.200	0.654	3
Lilit	0.017	0.018	0.958	4.815	>	3.400	0.696	2
Lipyg	0.016	0.015	1.046	1.111	<	1.400	0.735	1
Pavul	0.015	0.019	0.800	4.222	>	3.800	0.772	2
Chcri	0.013	0.014	0.941	1.370	>	0.600	0.804	1
Hasil	0.012	0.013	0.873	0.333	<	1.200	0.832	1
Elmod	0.011	0.017	0.671	0.556	<	0.800	0.859	2
Alesc	0.009	0.012	0.769	0.444	<	0.800	0.881	1
Lisax	0.009	0.016	0.553	0.963	>	0.000	0.902	2
Chste	0.009	0.015	0.559	0.259	<	0.800	0.923	2
Hapan	0.008	0.014	0.580	1.000	>	0.000	0.943	2
Gicin	0.007	0.011	0.598	0.778	>	0.000	0.960	2
Papal	0.004	0.009	0.413	0.482	>	0.000	0.969	1
Mener	0.003	0.008	0.332	0.370	>	0.000	0.976	2
Pauly	0.003	0.005	0.568	0.148	<	0.200	0.983	2
Sebal	0.002	0.006	0.397	4.889	<	5.000	0.989	2
Anvir	0.002	0.009	0.195	0.185	>	0.000	0.993	2
Salat	0.002	0.008	0.195	0.185	>	0.000	0.997	1
Hielo	0.001	0.007	0.195	0.148	>	0.000	1.000	1
Lahyp	0.000	0.000	NaN	0.000	=	0.000	1.000	1
Padep	0.000	0.000	NaN	0.000	=	0.000	1.000	2

Table A 2-8. SIMPER analysis on full ecological community between sedimentary and igneous lithologies in Wales. Species are listed in order of their contribution to dissimilarities between assemblages. Average= species contribution to between-group dissimilarity, sd= standard deviation of contribution, ratio= average:sd ratio, Sedimentary mean abundance, Igneous mean abundance, cum sum= ordered cumulative contribution and trophic level of species.

Wales- sedimentary-igneous								
Species	**Average**	**sd**	**ratio**	**Sed mean abu**		**Ign Mean abu**	**Cum sum**	**Trophic level**
Ladig	0.014	0.011	1.310	2.444	<	3.500	0.044	1
Elmod	0.014	0.010	1.373	3.857	>	2.000	0.088	2
Myspp	0.013	0.011	1.229	2.571	>	0.912	0.130	2
Baper	0.013	0.011	1.241	2.333	<	2.654	0.172	2
Mener	0.012	0.010	1.169	3.730	>	3.115	0.210	2
Pauly	0.012	0.009	1.332	2.159	<	2.850	0.248	2
Fuspi	0.012	0.010	1.215	3.048	<	3.231	0.285	1
Chste	0.012	0.009	1.355	2.589	<	2.769	0.322	2
Fuves	0.011	0.009	1.216	2.778	<	3.846	0.356	1
Lipyg	0.011	0.009	1.136	2.016	>	0.500	0.390	2
Gicin	0.010	0.007	1.471	1.714	<	2.231	0.423	2
Maste	0.010	0.009	1.177	2.968	<	3.962	0.456	1
Lahyp	0.010	0.009	1.139	1.413	<	1.500	0.488	1
Asnod	0.010	0.009	1.143	1.984	>	1.154	0.520	1
Oslin	0.010	0.008	1.320	2.794	<	2.846	0.552	2
Hielo	0.010	0.009	1.031	1.444	>	1.231	0.583	1
Lisax	0.010	0.010	1.001	4.270	>	3.885	0.614	2
Chcri	0.009	0.007	1.302	2.079	<	2.500	0.643	1
Anvir	0.009	0.008	1.201	1.730	>	1.077	0.673	2
Lilit	0.009	0.008	1.064	3.651	<	3.692	0.701	2
Padep	0.009	0.007	1.182	2.889	<	3.539	0.728	2
Pecan	0.009	0.007	1.171	3.587	<	3.692	0.755	1
Acequ	0.008	0.007	1.240	3.556	<	2.654	0.782	3
Fuser	0.008	0.010	0.873	3.857	<	4.769	0.808	1
Hapan	0.007	0.006	1.225	1.841	>	1.731	0.831	2
Samut	0.007	0.008	0.838	1.333	>	0.231	0.853	1
Nulap	0.007	0.006	1.179	3.810	>	3.577	0.875	3
Giumb	0.006	0.008	0.787	4.206	<	4.231	0.895	2
Salat	0.006	0.007	0.927	0.524	<	0.962	0.914	1
Chmon	0.006	0.008	0.793	4.476	>	4.423	0.934	2
Auver	0.006	0.006	0.950	0.952	>	0.500	0.951	3
Sebal	0.004	0.005	0.862	4.857	<	4.923	0.964	2
Bibif	0.003	0.007	0.529	0.714	>	0.000	0.975	1
Alesc	0.003	0.006	0.489	0.063	<	0.539	0.985	1
Pavul	0.002	0.004	0.574	4.810	<	4.846	0.992	2
Hasil	0.002	0.005	0.350	0.175	>	0.154	0.998	1
Saalv	0.001	0.004	0.179	0.127	>	0.000	1.000	2

Chapter 3 Appendix

Table A 3-1. GLM results for species richness (Quasi-Poisson), mobile species abundance (Negative Binomial) and sessile fauna abundance (Negative Binomial where tested) by feature type for each lithology. Where HC= hairline crack, DP= deep pool, SP= shallow pool and adj = adjacent (***=p<0.001, **=p<0.01, *=p<0.05).

Basalt				
Species richness (Quasi-Poisson)	**Estimate**	**Std. Error**	**Z value**	**P-value**
HC- DP == 0	-1.792	0.420	-4.264	***
Ledge- DP == 0	-0.962	0.258	-3.733	**
Ledge adj -DP == 0	-2.108	0.394	-5.355	***
Pool adj - DP == 0	-1.766	0.372	-4.743	***
Surface- DP == 0	-1.246	0.238	-5.240	***
Overhang - Ledge adj == 0	1.743	0.558	3.124	*
Mobile abundance (Negative Binomial)	**Estimate**	**Std. Error**	**Z value**	**P-value**
HC- DP == 0	-3.714	0.987	-3.763	**
Ledge adj -DP == 0	-2.169	0.674	-3.219	*
Surface - DP == 0	-2.797	0.630	-4.441	***
Overhang - HC == 0	4.848	1.380	3.513	*
Surface – Ledge == 0	-1.852	0.564	-3.286	*
Surface - Overhang == 0	-3.932	1.152	-3.413	*
Limestone				
Species richness (Quasi-Poisson)	**Estimate**	**Std. Error**	**Z value**	**P-value**
Surface- Crevice == 0	-2.197	0.434	-5.059	***
Ledge- DP == 0	-0.567	0.157	-3.609	**
Ledge adj - DP == 0	-2.628	0.473	-5.553	***
Pool adj - DP == 0	-1.906	0.269	-7.082	***
Surface- DP == 0	-2.405	0.326	-7.383	***
Ledge adj - Ledge == 0	-2.061	0.474	-4.345	***
Pool adj - Ledge == 0	-1.339	0.271	-4.940	***
Surface - Ledge == 0	-1.838	0.327	-5.615	***
SP - Ledge adj == 0	2.420	0.486	4.984	***
SP- Pool adj == 0	1.698	0.290	5.852	***
Surface- SP == 0	-2.197	0.343	-6.399	***
Mobile abundance (Negative-Binomial)	**Estimate**	**Std. Error**	**Z value**	**P-value**
Ledge adj -DP == 0	-3.368	0.650	-5.180	***
Pool adj - DP == 0	-1.952	0.486	-4.019	**
Surface- DP == 0	-2.164	0.484	-4.470	***
Ledge adj- Ledge == 0	-2.645	0.615	-4.302	***
Surface – Ledge == 0	-1.441	0.435	-3.309	*
Pit - Ledge adj == 0	3.108	0.917	3.391	*
SP – Ledge adj == 0	2.732	0.711	3.842	**
Sessile fauna abundance (count) (Negative Binomial)	**Estimate**	**Std. Error**	**Z value**	**P-value**
Ledge - DP == 0	-3.749	0.487	-7.697	***
Ledge adj - DP == 0	-6.003	1.125	-5.337	***

Pit - DP == 0	-3.413	0.929	-3.675	**
Pool adj - DP == 0	-4.454	0.574	-7.766	***
SP - Ledge == 0	2.218	0.567	3.913	**
SP - Ledge adj == 0	4.472	1.162	3.850	**
SP - Pool adj == 0	2.923	0.643	4.549	***
Andesite				
Species richness (Quasi-Poisson)	**Estimate**	**Std. Error**	**Z value**	**P-value**
Ledge - DP == 0	-0.829	0.227	-3.647	**
Ledge adj - DP == 0	-3.209	0.638	-5.029	***
Pool adj - DP == 0	-1.754	0.421	-4.170	***
Surface - DP == 0	-6.286	0.202	-3.115	*
Ledge adj - Ledge == 0	-2.380	0.642	-3.706	**
Overhang - Ledge adj == 0	2.890	0.800	3.614	**
SP - Ledge adj == 0	2.380	0.685	3.474	**
Surface - Ledge adj == 0	2.580	0.634	4.073	***
Mobile abundance (Negative Binomial)	**Estimate**	**Std. Error**	**Z value**	**P-value**
Ledge - DP == 0	1.492	0.511	2.920	*
Ledge adj - Ledge == 0	-3.401	0.586	-5.808	***
SP - Ledge == 0	-1.920	0.633	-3.032	*
Surface - Ledge == 0	-1.427	0.390	-3.661	**
Surface - Ledge adj == 0	1.974	0.577	3.420	**
Sandstone				
Species richness (Quasi-Poisson)	**Estimate**	**Std. Error**	**Z value**	**P-value**
Ledge adj - Crevice == 0	-1.910	0.550	-3.475	*
Ledge adj - DP == 0	-2.293	0.497	-4.613	***
Pool adj - DP == 0	-1.769	0.421	-4.202	***
Surface - DP == 0	-7.546	0.228	-3.311	*
Pit - Ledge adj == 0	1.591	0.494	3.222	*
SP - Ledge adj == 0	1.910	0.528	3.617	**
Surface - Ledge adj == 0	1.538	0.472	3.257	*
Mobile abundance (Negative Binomial)	**Estimate**	**Std. Error**	**Z value**	**P-value**
Surface - Crevice == 0	-2.901	0.647	-3.539	*
Pit - Ledge adj == 0	1.764	0.531	3.322	*
Surface - Pit	-1.836	0.399	-4.597	***

Table A 3-2. Kruskal-Wallis and Dunn's post-hoc results (with Bonferroni adjustments) for algae and lichen abundance Where HC= hairline crack, S= surface, L adj = ledge adjacent, P adj= pool adjacent, L= ledge DP= deep pool and SP= shallow pool (***=p<0.001,**=p<0.01).

Metric	**Lithology**	**df**	***H* (X^2)**	**P**	**Post-hoc adjustments**	**Comparison**	**Z-value**	**P**
Algae and lichen abundance	Basalt and Basaltic Andesite	9	32.542	***	P=0.05/45= 0.0011	DP > L DP > L adj DP > S	4.295 5.162 4.120	*** *** ***
	Limestone	7	72.423	***	P=0.05/28= 0.0018	DP > L DP > L adj DP > P adj DP > S L < SP P adj < SP SP > S	6.083 5.159 6.390 6.469 -3.980 -4.342 4.381	*** *** *** *** ** *** ***
	Andesite and Basalt	7	37.729	***	P= 0.05/28= 0.0018	DP > L DP > L adj DP > P adj	5.114 5.208 4.593	*** *** ***
	Sandstone	8	25.667	**	P=0.05/36= 0.0014	NS		

Table A 3-3. Presence of species in microhabitats (S=surface, HC= hairline crack, C= crack, Cr= crevice, P= pit, SP= shallow pool, DP= deep pool, PA= pool adjacent, L=ledge, LA=Ledge adjacent, O=overhang) by lithology (BL= Barns Ness limestone, BS= Barns Ness sandstone, G=Glamorgan Blue Lias limestone, A= Andesite, B=Basalt, S=sandstone, L=limestone). Where lithologies are recorded in microhabitats, species presence is indicated.

Species	S	HC	C	Cr	P	SP	DP	PA	L	LA	O	Most frequented habitats
Littorina littorea	BL, BS, L, S	B, G	G	B, BL, BS, L, S, G	L, S	B, G, BL, L, A, S	B, BL, L,A,S, BS	B, L, S	L, A,S	B,L,S	B	Ubiquitous but greatest in pools and crevices
Littorina saxatilis	BL			B, BS, G		G	B, BL, BS		G			Pools and crevices
Littorina obtusata				BS		L						Pools and crevices
Gibbula umbilicalis				G		G	A		S			Pools and crevices
Patella vulgata	B, BL, BS, A, S, G		B	BS, L,S,G	L, S	B,L,A, S, G	B,L,A, S	B, L, A,S, G	B, BL, L, A,S, G	B, BL, L,A,S	B,S, A	Ubiquitous
Patella depressa				G		G		G				Pools and crevices
Nucella lapillus				L, G								Crevices
Carcinus maenas							L					Pools
Ligia oceanica	BL, A			BS					B		S	No clear preference
Gammarus sp.	A, S				S						S	Pits
Actinia equina				L	L	B,L	B,L,A		B, L		B,A	Pools
Mytilus edulis		B	B	S			B,L	L	L	L	A	No clear preference
Polydora ciliata				G		L, G	L					Features that hold water-typically pools
Bdellidae sp.									L			Ledge

Anurida maritima					S	S					Pools
Fucus spiralis	BL, A, S, BS	BL	B	BS, S	S	S	B, BL, L,A,S		B, L, S		Surface
Fucus serratus							A				Pools
Fucus vesiculosus	S										Surface
Ulva sp.	B, BL, A, S			BL	S	B, BL, A,L	B, BL, L,A,S			S	Pools
Cladophora sp.	S			S			B,L,S, A			S	Pools
Rhodothamniella floridula				BS		B	B, BS				Pools
Chondrus crispus	B, A		G	L		L	L,A		B, L	L	Pools
Mastocarpus stellatus	S						S		S		Pools
Corallina officialis						L	B,L,A, S				Pools
Coralline crust (Lithothamnion sp.)							B,L,A, S				Pools
Pelvetia canaliculata	B, BL, BS, S	B	BS	BS		B	B	B	B, BL, A		Surface and ledge
Porphyra umbilicalis	S										Platform surface
Verrucaria sp.	S										Platform surface
Lichina pygmaea	B, BL, BS			BS			B, BL				Platform surface
Barnacle spp.	B, BL, L,A,S, G	B, G	B, G	BS, L,S	S	A, S	B	B, L, G	B, L, A,S, G		Depends on lithology not habitat type

Table A 3-4. Negative Binomial GLM significant post-hoc comparisons for Gastropoda abundance combined for all rocky shore survey data (***=p<0.001, **=p<0.01, *=p<0.05).

Feature Comparison	**Estimate**	**Std. Error**	**Z value**	**P**
Crevice – Crack == 0	3.873	0.830	4.667	***
Deep pool- Crack == 0	4.327	0.831	5.205	***
Ledge- Crack == 0	3.999	0.819	4.885	***
Overhang- Crack == 0	4.848	1.035	4.685	***
Pit- Crack == 0	4.303	0.893	4.821	***
Pool adjacent- Crack == 0	2.981	0.826	3.610	*
SP- Crack == 0	3.288	0.835	3.940	**
Hairline crack- Crevice == 0	-3.350	0.587	-5.692	***
Ledge adjacent – Crevice == 0	-1.945	0.340	-5.727	***
Surface- Crevice == 0	-1.993	0.253	-7.871	***
Hairline crack- Deep pool == 0	-3.793	0.589	-6.445	***
Ledge adjacent – Deep pool == 0	-2.399	0.343	-6.998	***
Pool adjacent- Deep pool == 0	-1.346	0.305	-4.412	***
Shallow pool- Deep pool == 0	-1.039	0.328	-3.166	*
Surface- Deep pool == 0	-2.447	0.257	-9.508	***
Ledge- Hairline crack == 0	3.465	0.571	6.074	***
Overhang- Hairline crack == 0	4.315	0.852	5.062	***
Pit- Hairline crack == 0	3.770	0.673	5.605	***
Pool adjacent- Hairline crack == 0	2.447	0.581	4.214	***
Shallow pool- Hairline crack == 0	2.755	0.593	4.643	***
Ledge adjacent- Ledge == 0	-2.071	0.311	-6.662	***
Pool adjacent- Ledge == 0	-1.018	0.269	-3.788	**
Surface- Ledge == 0	-2.119	0.213	-9.949	***
Overhang- Ledge adjacent == 0	2.920	0.705	4.140	**
Pit- Ledge adjacent == 0	2.375	0.473	5.025	***
Pool adjacent- Ledge adjacent == 0	1.053	0.329	3.198	*
Shallow pool- Ledge adjacent == 0	1.360	0.351	3.878	**
Surface- Overhang == 0	-2.968	0.668	-4.443	***
Surface- Pit == 0	-2.423	0.415	-5.839	***
Surface – Pool adjacent == 0	-1.101	0.239	-4.605	***
Surface- Shallow pool == 0	-1.408	0.268	-5.256	***

Chapter 4 Appendix

Table A 4-1. Example ecological studies- tiles panels and settlement plates (n=61). N= number of sites in study.

	Author	Location	N	Location on shore/off shore	Enhancement Material	Enhancement Design	Size	Replicates
1.	(Field et al., 2007) (*Sub-study*)	Eilat Marine Laboratory, Israel	1	Reef	Unglazed Ceramic	Tiles- Relatively smooth surface	100x100x5 mm	**3** of each material on wire racks
						Tiles-Pitted surface		
					Brick Fired		115x115x25 mm	9 of each material attached to substrate
2.	Field et al., 2007) (*Study*)	Eilat, Israel	2	Fore reef slope	Ceramic	Tiles- Variations in tile size tested	200x200 mm 200x100 mm 100x100 mm	**4** of each size per wire rack
		Straits of Tiran, Egypt		Fore reef/Back reef slope	Masonry (Brick Fired)		220x110 mm 110x110 mm 55x110 mm	Approximately 9 of each size per rack
3.	(Fisk and Harriott, 1990)	Sharm el Sheikh Green, Michaelmas and Upolu Reefs, Australia	3	Fringing reef, seaward slope	Ceramic	Tiles	15x15 cm	**4** per rack
				Mid-shelf reef-Fore reef/back reef slope				**4** per rack
4.	(Coombes et al., 2015)	Cornwall, UK	2	Mean tide level (MTL)	Marine grade concrete	Tiles (control, smoothed, grooved, exposed aggregate)	5 cm x5 cm x3 cm	**10** replicates

5.	(Loke et al., 2014)	South of Singapore Island	2	Low-shore height at two islands	Concrete Granite control tile	Tiles- 2 designs of varying structural complexity but equal surface area (8-56 mm scale)	40x40x6 cm^3	**5** of each tile type
6.	(Baird and Hughes, 2000)	Lizard Island, northern Great Barrier Reef	1	Mid-shelf reef at 2 m depth	Unglazed clay	Paving tiles sandwiched together around mesh so that tiles split into ‘upper’/‘lower’	11x11x1 cm	**6** tiles per rack
7.	(Perkol-Finkel and Sella, 2014)	Red Sea, Egypt Mediterranean Sea, Ashdod, Israel	2	10 m depth 6 m depth	Concrete- five different concrete compositions	Tiles- One smooth face, one textured	15x15x4 cm	**10** replicates for each matrix and control
8.	(Maida et al., 1995)	Orpheus Island, Great Barrier Reef Lizard Island, Great Barrier Reef	2	Southern fringing reef, Pioneer Bay, mean depth 4 m Western fringing reef, Palfrey Island, mean depth 4 m	Ceramic	Tiles- Arranged into settlement stacks (only used to highlight coral growth direction and distance)	15x15 cm	**5** tiles for each settlement stack
9.	(Fletcher et al., 2013)	Nelson, New Zealand Ruakaka Bay, New Zealand	2	Floating arrays, tiles positioned between 0.5-1.5 m	Black Perspex	Roughened settlement plates	20x20 cm	**15** settlement plates at each site 3 plates per array

10.	(Burt et al., 2009)	Dubai, UAE	4	2 natural reef and 2 breakwater reef sites ≈ 4 m depth	Concrete Gabbro Granite Sandstone Terra-cotta	Tiles	100x100x14 mm	**25** tiles of each material at each site
11.	(Hughes et al., 2002)	Great Barrier Reef, Australia	132	33 reefs	Unglazed clay	Tiles	11x11x1 cm	**10** recruitment panels per site
12.	(Chabot and Bourget, 1988) *Study 1*	New Brunswick, Canada	2	Mid-intertidal zone	Slate	Panels- artificial crevices	45x30 cm (crevice 26 cm deep, angled 60 degrees at base)	**8** artificial crevices created
13.	(Chabot and Bourget, 1988) *Study 2*	New Brunswick, Canada	2	Mid-intertidal zone	Slate	Ungrooved and grooved (5 mm/10 mm) panels	15x15 cm	**12** panels attached to boards at each location
14.	(Mundy, 2000)	Heron Reef, Great Barrier Reef	2	North slope of reef ≈9 m depth	Terracotta	Settlement plate with pits and grooves (1 mmx1 mm)	110 mmx110 mmx10 mm	**10** replicates at each site
15.	(Lathlean and Minchinton, 2012)	Garie Beach, Sydney, Australia	1	Mid-intertidal heights	PVC plates	Black Grey White	10x10x0.5 cm	**5** plates of each treatment
16.	(Menge et al., 2010)	Central Oregon coast	5	Lower end of mid-intertidal zone	PVC plates	Saf-T-Walk tape coated plates PVC plates Travertine plates (low-moderate texture) Natural Rock	10x10 cm	**5** replicates of eight treatments

17.	(Pineda and Caswell, 1997)	Laboratory setting	-	-	Ceramic	Grooved tiles	11.2x11.2 cm	**5** replicates of each treatment
18.	(Villamagna and Strayer, 2009)	Mid-Hudson	4	River- 0.5 m below low water mark	Concrete	Tiles- from smooth to rough (crevices and peaks exaggerated)	9.92 cm x9.92 cm	**10** of each surface roughness at each site
19.	(Raimondi, 1990)	Punta Pelicano, Gulf of California	1	Exposed reef	Plexiglass	Plates- Uniformly pitted 2.4 mm deep, 2 mm diameter	7x7 cm	**8** replicates
20.	(Johnson and Strathmann, 1989)	San Juan Island, Washington	1	Intertidal zone	Slate	Tiles- 12-15 horizontal and vertical grooves on tiles (1 mmx1 mm)	14.5x14.5x0.8 cm	**12** replicates 6 pairs of plates
21.	(Lozano-Cortés and Zapata, 2014))	Gorgona Island, Colombia	1	3 reef zones - backreef, flat and slope reef zones	Terracotta Ceramic	Plates	20x20x0.5 cm 20x20x1.0 cm	**5** of each substrate in each reef zone
22.	(Bulleri, 2005)	Sydney Harbour, Australia	3	Seawall and vertical rocky shore within a few metres 0.9-0.7 m above mean low water	Sandstone	12 clearings or sandstone panels produced/fixed	13x13x3 cm	**4** replicates of each substratum type (clearings vs panels)
23.	(Glasby and Connell, 2001)	Sydney Harbour, Australia	3	Rocky reef Pontoons (≈25 cm depth)	Concrete	Settlement panels-Oriented horizontally and vertically	15x15 cm	**5** per orientation per site
24.	(Hills and Thomason, 1998)	Millport, Clyde Sea, UK	1	Mid-intertidal	Filled-Polyester	Tiles- Four textures- Smooth,	20x20 cm (93x93 mm with a smooth	**5** replicates of each texture

						fine, medium and coarse	edge area of 10 mm for attachment)	3 replicates of each size of tile
						Smooth textured tiles of 5 different sizes were also made	1 x 1 cm 2 x 2 cm 5 x 5 cm 10 x 10 cm 20 x 20 cm	
25.	(Thomason et al., 2002)	Straits of Tiran, Red Sea, Egypt	1	Fringing reef slope at 5 m depths	Carbonate filled polyester resin Epoxy Silanised epoxy resin	Tiles- Five textures from smooth to rough (mean roughness 0 mm to 2.18 mm)	100x100x5 mm	**8** replicates of each tile Each texture/resin combination replicated 4 times each side
26.	(Todd et al., 2006)	Fife, Scotland	6	Barnacle zone on the shore at the height of the highest neap ebb tide	Ceramic Acrylic	Tiles- 58 (≈ 1 mm deep) depressions Panels- Ten (1 mm deep horizontally and 0.5 mm deep vertical) grooves machined	9.8x9.8 cm 7x7x0.6 cm	**4** replicates **3** replicates
27.	(Thomason et al., 2000)	Millport, Clyde Sea, UK	1	Mid-point of barnacle zone (equivalent)	Filled-polyester resin	Tiles- fine surface texture (<0.5 mm diameter grain size)	9.3x9.3 cm	**6** replicates
28.	(McCulloch and Shanks, 2003)	Sunset Bay, Oregon	1	0.5 m below surface	Plexiglass	Plates coated with Safety-Walk tape	10x10 cm	**4** per mooring line

29.	(Petraitis, 1991)	Maine, USA	6	Mean low water (between -0.2 and +0.2 m)	Fiberglass resin	Plates cast from moulds of natural barnacle aggregations	200 cm^2	**8** replicates of 9 different moulds
30.	(Leslie et al., 2005)	Oregon, USA	6	Mid-intertidal zone (2 m above MLLW)	PVC	Plates- pitted (1 mm diameter, 0.3 mm deep), 81 pits/plate	10x10 cm 6.35 mm thick	**26-27** replicates
31.	(Pineda and López, 2002)	Alta and Baja, California	2	Mid-intertidal zone	PVC	Plates- halved PVC pipes, three sharp grooves machined into plates	11 cm long ≈ 2.5 cm inner diameter	**3-9** plates
32.	(Connell, 2000)	Sydney Harbour, Australia	4	25 cm depth	Sandstone	Plates- Grain size same as subtidal rock	15x15 cm	**5** of each type in each habitat
				1.5 m below low water spring tide	Concrete	Plates- Smooth texture		
33.	(Lagos et al., 2005)	Central Chile coastline	16	Mid and low intertidal zones	Plexiglas	Plates coated with SWT	10x10 cm	**4**
34.	(Shanks, 2009a)	Oregon, USA	1	Barnacle zone of the intertidal	Safety Walk Tape	Plates	20x20 cm (Divided into 5x5 cm quarters)	**3**
35.	(Bell et al., 2015)	Plettenberg Bay, South Africa	2	Mid mussel zone	Perspex	Plates- Four surfaces (live mussels, mussel shells only, resin shells that mimic shell micro-surface and rock mimic)	8.0x5.5 cm	**11** of each treatment

36.	(Menge et al., 1999)	South Island, New Zealand	4	Upper and lower levels of barnacle and mussel zones	Plexiglas	Plates- Coated with STW tape	10x10 cm 5 mm thick	**5** per zone per site
37.	(Bowden et al., 2006)	Adelaide Island, Antarctic Peninsula	3	8 m and 20 m depths	Acrylic	Plates were roughened black acrylic sheet	250x150x5 mm	**6** settlement plate unit at each depth at each site
38.	(Butler, 1986)	Gulf St. Vincent, South Australia	5	At least 1.5 m above seafloor, 4 m below low water mark	"Hardiflex" asbestos cement sheet	Plates- one smoother side	15x12 cm ≈ 5 mm thick	**10** within a pier at a time
39.	(Johnston et al., 2002)	Port Shelter, Hong Kong	1	>3.5 m depth	Perspex	Plate- roughened by sanding	11x11x1 cm	**4-8** (for each treatment)
40.	(Dunstan and Johnson, 1998)	Heron Reef, Great Barrier Reef	9	Reef slopes (9-12 m depth) and lagoonal coral bommies (2-3 m depth)	Ceramic	Unglazed surfaces – bathroom tiles	200x200 mm	**20** plates at each site
41.	(Blum et al., 2007)	San Francisco Bay, US	1	1 m depth	PVC	Panels- sanded	13.7x13.7 cm 0.25 cm thick	**5**
42.	(Blum et al., 2007) *Sub study*	San Francisco Bay, US	1	1 m depth	PVC	Four different sizes	7x7 cm (small) 13.7x13.7 cm (medium) 21.7x21.7 (large) 34.3x34.3 cm (extra-large)	**10** of each size and of three potential treatments
43.	(Johnston and Keough, 2000)	Port Phillip Bay, Victoria, Australia	2	3.5 m below the low water mark	Plexiglas	Plates- black Perspex	11x11x1 cm	**4** per treatment

44.	(Harriott and Banks, 1995)	Solitary Islands, Australia	4	Three islands (3-11km offshore) and a rocky reef near the mainland, depth 6-9 m	Ceramic	Plates	15x15 cm	**8** plates per rack
45.	(Moschella et al., 2005)	Elmer defence scheme, UK	2	Mid-tide level	Concrete	Panels- varying levels of complexity from smooth to pitted of varying size	30x30 cm	**4** per treatment and orientation
46.	(Ferrario et al., 2016)	Marotta, Cesenatico and Punta Marina, Italy	3	Landward of breakwaters	Marble	Tiles of three treatments	10x10x2 cm	**5** for each treatment
47.	(Porzio et al., 2013)	Ischia, Tyrrhenian Sea, Italy	6	1-2 m depth, rocky subtidal	Volcanic stone	Tiles- Volcanic stone surface comparable to adjacent rocky subtidal	15x15x1 cm	**9** per site
48.	(Doropoulos et al., 2014)	Palau, Western Micronesia	6	Forereefs, 7 m depth	Ceramic	Tile- fine manufactured rugosity on surface	5x5 cm	**15** tile pairs per reef
49.	(Sneed et al., 2014)	Looe Key, Florida	1	Near Looe Key, Florida	Limestone	Tiles- prepared with biofilms either 15 or 21 days old	5x5x1.2 cm	**8** replicates per treatment
50.	(Hepburn et al., 2015)	Puerto Morelos reef, Mexico	5	Lagoon, Back reef- , Reef crest- , Reef front- 5 m depth	Ceramic	Tiles- Changing orientation of the blocks that tiles were attached to	15x20 cm^2	**9** replicates from six microhabitats

51.	(Blakeway et al., 2013)	Dampier Harbour, Western Australia	12	Reef front- 8 m depth Six rock substrate sub sites across the artificial reef and six at the natural reef	Terracotta	Tiles	11x11x1 cm	**12 per subsite**
52.	(Stubler et al., 2016)	Jamaica, West Indies	3	8-10 m depth and 15-18 m depth	Terracotta	Tiles	12x12x1 cm	**9** per site and depth
53.	(Taniguchi and Tokeshi, 2004)	Amakusa Shimoshima Island, southern Japan	1	Along a 150 m stretch of stream	Serpentinite stone	Plates- five levels of complexity attached to a basal plate- total surface area always (20x20x1 cm)	No alterations 10x10 cm 5x5 cm 2.5x2.5 cm 1x1 cm (all 0.3 cm thick)	**12** replicates for each complexity treatment
54.	(Plass-Johnson et al., 2016)	Spermonde Archipelago, Indonesia	3	Fore-reef, 5 m depth	Granite	Tiles (three different cage treatments)	10x10 cm 1.5 cm thick	**8** tiles per cage, three replicates of each cage treatment per site
55.	(Jenkins et al., 1999)	Isle of Man, UK	3	Rocky intertidal, high, mid and low shore	Slate	Tiles- natural, barnacle encrusted rock	13x13 cm (minimum)	**5** replicates (average of several experiments)
56.	(Herbert and Hawkins, 2006)	Lyme Regis, Dorset, UK	1	Flat shore platform, mean tide level	Bembridge Limestone (Isle of Wight) Chalk (Isle of Wight) Kimmeridge Cementstone	Tiles-one face remains naturally weathered	6x4x1.5 cm	**10** replicates for each rock type

					(Dorset) Blue Lias Limestone (Dorset)			
57.	(Prendergast et al., 2008)	Millport, Clyde Sea, UK	1	Mid barnacle zone	Carbonate filled polyester resin in styrene	Tiles- 10x10 cm textured with smooth resin border Tile textures- Smooth, very fine, fine, coarse and tough	12x12 cm	**10** replicates of each texture
58.	(Shanks, 2009b)	Bastendorff Beach, Oregon	1	≈1.75 m above zero tide level	Plexiglas	Plates- coated in grey Safety Walk tape	10x10 cm	**3** /orientation
59.	(Shanks, 2009b) *Sub-study*				Ceramic	Tiles- unglazed brown and grey, grooved bottom surface Tiles- white, glazed top and grooved bottom	15x15 cm 1 cm thick 11x11 cm 0.7 cm thick	
60.	(Prendergast et al., 2009)	Millport, Clyde Sea, UK	1	25 m north of Keppel pier, attached , mid-barnacle zone	Carbonate filled polyester resin in styrene	Panels/Tiles- Double sided Textures from smooth to rough	12x12 cm^2	**5** replicates of each texture
61.	(Ellrich et al., 2015)	Deming Island, Nova Scotia, Canada	1	Rocky intertidal, 1.8 m elevation	PVC	Tile- coated in a black tape with sandpaper texture Four treatments	8.9x4.6x0.35 cm	**12** replicates per treatment

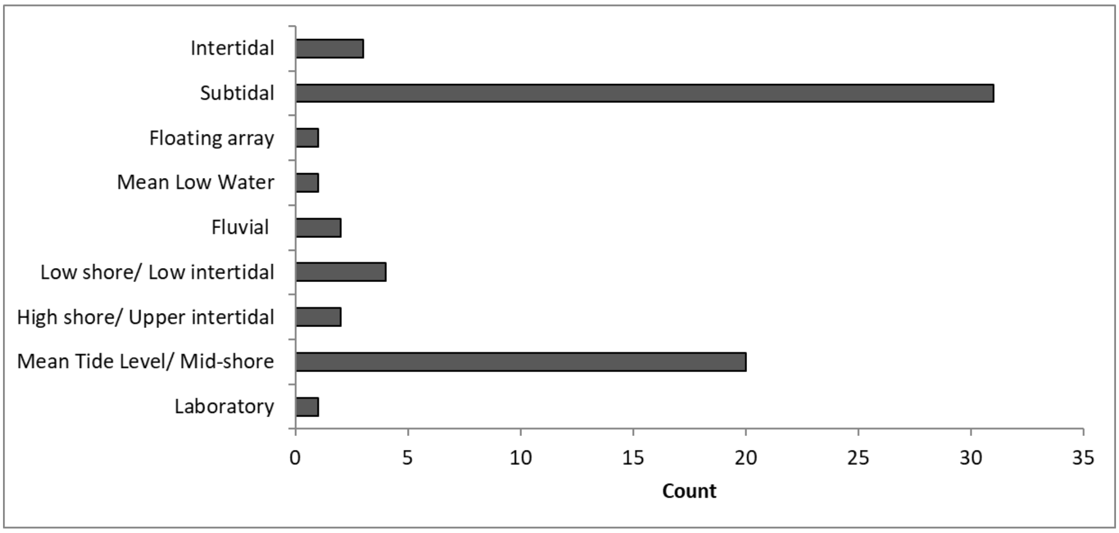

Figure A 4-1. Location of ecological studies from Table A 4-1- tiles panels and settlement plates, where n=65 from 61 studies.

Table A 4-2. Details of geotile mesh.

Mesh Bounding Box Size 0.819235 1.287842 1.287842 Mesh Bounding Box Diag 1.997053 Mesh Surface is 0.689598 Mesh Total Len of 1621701 Edges is 2055.481689 Avg Len 0.001267 Mesh Total Len of 1621701 Edges is 2055.481689 Avg Len 0.001267
Mesh has 1080460 triangles 0 quads 1080460 polygons 1080460 large polygons (with internal faux vertexes) Right Angle Discrepancy Avg 1.#QO Min 147.296 Max 147.296 StdDev 1.#QO Percentile 0.05 1.#QO percentile 95 1.#QO Quad Ratio Avg 0.000 Min 0.000 Max 0.000

Table A 4-3. Models selected for a) Barnacle cover b) Species richness (mobile and algae) and c) Mobile species abundance (df =degrees of freedom, AIC= Akaike's Information Criterion). Transformations of data are listed in fixed effects.

a) Barnacle cover **Saltcoats**				
Model: gls(log(Barnaclepercent+1)~Months.After.Deployment*TileType, weights=varConstPower(form =~Months.After.Deployment|TileType), data=data1)				
Source	**df**	**AIC**	**Log-likelihood**	**Post-hoc df**
log(Barnaclepercent+1)~ Months.After.Deployment*TileType	33	761.2160	-347.6080	215
Isle of Wight				
Model: lme(Barnaclepercent~Months.After.Deployment*TileType, weights=varPower(form =~Months.After.Deployment|TileType), random=~1|factor(rep), data=data2, control=lmeControl(opt='optim',maxIter = 1e8, msMaxIter = 1e8))				
Source	**df**	**AIC**	**Log-likelihood**	**Post-hoc df**
Barnaclepercent~Months.After. Deployment*TileType	26	1830.038	-889.0191	219
Blackness				
Model: lme(Barnaclepercent~log(Months.After.Deployment)*TileType+I(log(Months.After. Deployment)^2), weights=varPower(form =~Months.After.Deployment|TileType), random=~1|factor(rep), data=data3, control=lmeControl(opt='optim',maxIter = 1e8, msMaxIter = 1e8))				
Source	**df**	**AIC**	**Log-likelihood**	**Post-hoc df**
Barnaclepercent~log(Months.After.Deployment)*TileType+I(log(Months.After.Deployment)^2)	24	1634.345	-793.1724	196
b) Species richness (mobile and algal) **Saltcoats**				
Model: gls((SR.nobarn.)~Months.After.Deployment*TileType, weights=varExp(form =~Months.After.Deployment), data=data1)				
Source	**df**	**AIC**	**Log-likelihood**	**Post-hoc df**
(SR.nobarn.)~Months.After.Deployment*TileType	18	610.4098	-287.2049	215
Isle of Wight				
Model:				

gls(log(SR.nobarn.+1)~Months.After.Deployment*TileType, weights=varExp(form =~Months.After.Deployment|TileType), data=data2)				
Source	**df**	**AIC**	**Log-likelihood**	**Post-hoc df**
log(SR.nobarn.+1)~Months.After.Deployment*TileType	25	250.4215	-100.2107	226
Blackness				
Model: gls(SR.nobarn.~log(Months.After.Deployment)*TileType+I(log(Months.After.Deployment)^2), weights=varPower(form =~Months.After.Deployment), data=data3)				
Source	**df**	**AIC**	**Log-likelihood**	**Post-hoc df**
SR.nobarn.~log(Months.After.Deployment)*TileType+I(log(Months.After.Deployment)^2)	17	483.7781	-224.8891	203
c) Mobile species abundance Saltcoats				
Model: gls(log(Mobile+1)~Months.After.Deployment*TileType, weights=varConstPower(form =~Months.After.Deployment|TileType), data=data1)				
Source	**df**	**AIC**	**Log-likelihood**	**Post-hoc df**
log(Mobile+1)~Months.After.Deployment*TileType	33	372.4524	-153.2262	215
Isle of Wight				
Model: gls(log(Mobile+1)~Months.After.Deployment*TileType, weights=varConstPower(form =~Months.After.Deployment|TileType), data=data2)				
Source	**df**	**AIC**	**Log-likelihood**	**Post-hoc df**
log(Mobile+1)~Months.After.Deployment*TileType	33	507.6304	-220.8152	226
Blackness				
Model: gls(log(Mobile+1)~log(Months.After.Deployment)*TileType+I(log(Months.After.Deployment)^2), weights=varConstPower(form =~Months.After.Deployment|TileType), data=data3)				
Source	**df**	**AIC**	**Log-likelihood**	**Post-hoc df**
log(Mobile+1)~log(Months.After.Deployment)*TileType+I(log(Months.After.Deployment)^2)	30	471.1569	-205.5785	203

Table A 4-4. Summary of significant post-hoc comparisons for barnacle cover (%) between tile types by months after installation at each site. A GLS model was used for Saltcoats and LME models were used for the Isle of Wight and Blackness (MacArthur et al., 2019) (***=p<0.001, **=p<0.01, *=p<0.05).

Site/ Month	Comparison	SE	df	T ratio	Direction	P value
Saltcoats 2	Art 2- Control	0.276	215	6.894	>	***
	Art 2- Geotile	0.364	215	4.091	>	**
	Art 2- Singapore	0.276	215	7.029	>	***
	Art 3- Control	0.377	215	5.582	>	***
	Art 3- Geotile	0.445	215	3.790	>	**
	Art 3- Singapore	0.376	215	5.675	>	***
	Clearing- Control	0.460	215	3.879	>	**
	Clearing-Singapore	0.460	215	3.952	>	**
	Control-Grooved	0.356	215	-4.624	<	***
	Grooved-Singapore	0.355	215	4.722	>	***
Saltcoats 6	Art 2- Control	0.219	215	9.025	>	***
	Art 2- Geotile	0.216	215	4.979	>	***
	Art 2- Singapore	0.216	215	9.075	>	***
	Art 3- Control	0.294	215	6.945	>	***
	Art 3- Geotile	0.341	215	4.262	>	***
	Art 3- Singapore	0.292	215	6.936	>	***
	Barnacle- Control	0.358	215	3.882	>	**
	Barnacle- Singapore	0.356	215	3.847	>	**
	Clearing- Control	0.347	215	4.552	>	***
	Clearing- Singapore	0.344	215	4.521	>	***
	Control- Grooved	0.274	215	-6.468	<	***
	Geotile- Grooved	0.324	215	-3.654	<	**
	Grooved- Singapore	0.271	215	6.458	>	***
Saltcoats 12	Art 2- Control	0.293	215	7.127	>	***
	Art 2- Geotile	0.396	215	3.125	>	*
	Art 2- Singapore	0.277	215	7.182	>	***
	Art 3- Control	0.410	215	4.771	>	***
	Art 3- Singapore	0.398	215	4.657	>	***
	Barnacle- Control	0.451	215	3.162	>	*
	Clearing- Control	0.406	215	3.120	>	*
	Control- Grooved	0.347	215	-5.655	<	***
	Grooved- Singapore	0.334	215	5.585	>	***
Saltcoats 18	Art 2- Control	0.469	215	4.688	>	***
	Art 2- Singapore	0.443	215	4.564	>	***
	Control- Grooved	0.560	215	-3.851	<	**
	Grooved- Singapore	0.538	215	3.675	>	**
Site	Comparison	SE	df	T ratio	direction	P value
IOW 2	Art 1- Art 3	3.195	219	-5.114	<	***
	Art 1- Clearing	3.117	219	-3.976	<	**
	Art 1- Control	3.712	219	5.431	>	***

	Art 1- Grooved	3.307	219	-4.077	<	**
	Art 2- Control	3.569	219	7.989	>	***
	Art 2- Geotile	2.988	219	5.226	>	***
	Art 3- Barnacle	3.251	219	4.151	>	**
	Art 3- Control	3.668	219	9.950	>	***
	Art 3- Geotile	3.107	219	7.595	>	***
	Barnacle- Control	3.759	219	6.119	>	***
	Barnacle- Geotile	3.214	219	3.144	>	*
	Barnacle- Grooved	3.361	219	-3.165	<	*
	Clearing- Control	3.601	219	9.040	>	***
	Clearing- Geotile	3.028	219	6.492	>	***
	Control- Geotile	3.635	219	-3.548	<	*
	Control- Grooved	3.755	219	-8.933	<	***
	Geotile- Grooved	3.222	219	-6.437	<	***
IOW 6	Art 1- Art 2	2.303	219	-5.757	<	***
	Art 1- Art 3	2.420	219	-7.423	<	***
	Art 1- Barnacle	2.525	219	-3.427	<	*
	Art 1- Clearing	2.348	219	-4.853	<	***
	Art 1- Control	3.009	219	7.209	>	***
	Art 1- Grooved	2.533	219	-7.864	<	***
	Art 2- Control	2.877	219	12.150	>	***
	Art 2- Geotile	2.214	219	6.433	>	***
	Art 3- Barnacle	2.478	219	3.758	>	***
	Art 3- Control	2.972	219	13.343	>	***
	Art 3- Geotile	2.335	219	8.115	>	***
	Barnacle- Control	3.058	219	9.923	>	***
	Barnacle- Geotile	2.444	219	3.944	>	**
	Barnacle- Grooved	2.589	219	-4.353	<	***
	Clearing- Control	2.914	219	11.355	>	***
	Clearing- Geotile	2.261	219	5.477	>	***
	Clearing- Grooved	2.417	219	-3.526	<	*
	Control- Geotile	2.943	219	-7.035	<	***
	Control- Grooved	3.065	219	-13.577	<	***
	Geotile- Grooved	2.452	219	-8.525	<	***
IOW 12	Art 1- Art 2	3.248	219	-6.346	<	***
	Art 1- Art 3	3.321	219	-6.141	<	***
	Art 1- Barnacle	3.571	219	-4.862	<	***
	Art 1- Clearing	3.168	219	-3.124	<	*
	Art 1- Control	4.854	219	4.944	>	***
	Art 1- Grooved	3.592	219	-8.232	<	***
	Art 2- Clearing	2.675	219	4.006	>	**
	Art 2- Control	4.547	219	9.811	>	***
	Art 2- Geotile	2.764	219	4.409	>	***
	Art 3- Clearing	2.758	219	3.806	>	**
	Art 3- Control	4.603	219	9.645	>	***
	Art 3- Geotile	2.845	219	4.208	>	**
	Barnacle- Control	4.786	219	8.641	>	***
	Barnacle-Grooved	3.492	219	-3.496	<	*
	Clearing- Control	4.493	219	7.543	>	***

	Clearing- Grooved	3.079	219	-6.389	<	***
	Control – Geotile	4.547	219	-7.130	<	***
	Control - Grooved	4.802	219	-11.156	<	***
	Geotile- Grooved	3.157	219	-6.698	<	***
IOW	Art 1- Art 2	5.441	219	-5.140	<	***
18	Art 1- Art 3	5.501	219	-4.151	<	**
	Art 1- Barnacle	5.909	219	-4.413	<	***
	Art 1- Control	7.998	219	3.288	>	*
	Art 1- Geotile	5.374	219	-3.320	<	*
	Art 1- Grooved	5.943	219	-6.600	<	***
	Art 2- Clearing	4.426	219	4.420	>	***
	Art 2- Control	7.483	219	7.252	>	***
	Art 3- Clearing	4.489	219	3.215	>	*
	Art 3- Control	7.529	219	6.525	>	***
	Art 3- Grooved	5.283	219	-3.103	<	*
	Barnacle- Clearing	4.980	219	3.549	>	*
	Barnacle- Control	7.832	219	6.687	>	***
	Clearing- Control	7.347	219	4.723	>	***
	Clearing- Grooved	5.021	219	-6.139	<	***
	Control- Geotile	7.436	219	-5.935	<	***
	Control- Grooved	7.858	219	-8.339	<	***
	Geotile- Grooved	5.151	219	-4.152	<	**
Site	Comparison	SE	df	T ratio	direction	P value
Blackness	Art 1- Barnacle	6.503	196	-8.156	<	***
2	Art 1- Clearing	7.606	196	-6.736	<	***
	Art 1- Geotile	6.730	196	-6.588	<	***
	Art 1- Grooved	6.663	196	-9.714	<	***
	Barnacle- Control	6.606	196	6.618	>	***
	Barnacle- Singapore	7.366	196	8.822	>	***
	Clearing- Control	7.700	196	5.444	>	***
	Clearing- Singapore	8.343	196	7.574	>	***
	Control- Geotile	6.832	196	-5.126	<	***
	Control- Grooved	6.765	196	-8.189	<	***
	Geotile- Grooved	4.759	196	-4.282	<	***
	Geotile- Singapore	7.562	196	7.443	>	***
	Grooved- Singapore	7.504	196	10.216	>	***
Blackness	Art 1- Barnacle	3.369	196	-11.137	<	***
6	Art 1- Clearing	3.947	196	-7.488	<	***
	Art 1- Geotile	3.479	196	-9.285	<	***
	Art 1- Grooved	3.447	196	-12.580	<	***
	Art 1- Singapore	4.902	196	3.729	>	**
	Barnacle- Control	3.376	196	8.494	>	***
	Barnacle- Singapore	4.077	196	13.684	>	***
	Clearing- Control	3.947	196	5.248	>	***
	Clearing- Grooved	2.941	196	-4.695	<	***
	Clearing- Singapore	4.560	196	10.492	>	***
	Control- Geotile	3.484	196	-6.735	<	***

	Control- Grooved	3.452	196	-10.000	<	***
	Control- Singapore	4.901	196	5.534	>	***
	Geotile- Grooved	2.249	196	-4.917	<	***
	Geotile- Singapore	4.166	196	12.141	>	***
	Grooved- Singapore	4.140	196	14.889	>	***
Blackness	Art 1- Barnacle	3.665	196	-7.565	<	***
12	Art 1- Geotile	3.725	196	-6.635	<	***
	Art 1- Grooved	3.706	196	-8.066	<	***
	Art 1- Singapore	5.809	196	3.834	>	**
	Barnacle- Clearing	2.261	196	5.239	>	***
	Barnacle- Control	3.247	196	5.908	>	***
	Barnacle- Singapore	4.620	196	10.823	>	***
	Clearing- Geotile	2.357	196	-3.750	<	**
	Clearing- Grooved	2.326	196	-6.024	<	***
	Clearing- Singapore	5.047	196	7.560	>	***
	Control- Geotile	3.314	196	-4.880	<	***
	Control- Grooved	3.292	196	-6.484	<	***
	Control- Singapore	5.558	196	5.545	>	***
	Geotile- Grooved	1.310	196	-3.950	<	**
	Geotile- Singapore	4.667	196	10.067	>	***
	Grooved- Singapore	4.652	196	11.213	>	***
Blackness	Art 1- Barnacle	4.733	196	-4.648	<	***
18	Art 1- Geotile	4.801	196	-4.223	<	***
	Art 1- Grooved	4.778	196	-4.606	<	***
	Art 1- Singapore	7.408	196	3.322	>	*
	Barnacle- Clearing	2.862	196	4.933	>	***
	Barnacle- Control	4.127	196	3.303	>	*
	Barnacle- Singapore	5.858	196	7.957	>	***
	Clearing- Geotile	2.980	196	-4.160	<	***
	Clearing- Grooved	2.942	196	-4.804	<	***
	Clearing- Singapore	6.398	196	5.078	>	***
	Control- Grooved	4.181	196	-3.263	<	*
	Control- Singapore	7.048	196	4.679	>	***
	Geotile- Singapore	5.914	196	7.590	>	***
	Grooved- Singapore	5.895	196	7.908	>	***

Table A 4-5. Summary of significant post-hoc comparisons for species richness (mobile and algae, excluding barnacles) between tile types by months after installation at each site. GLS models were used for all sites (***=p<0.001, **=p<0.01, *=p<0.05).

Site	Comparison	SE	df	T ratio	Direction	P value
Saltcoats 2	Art 2- Barnacle	0.271	215	3.700	>	**
	Art 2- Control	0.271	215	3.175	>	*
	Art 2- Geotile	0.274	215	3.862	>	**
Saltcoats 6	Art 2- Art 3	0.208	215	3.357	>	*
	Art 2- Barnacle	0.200	215	4.722	>	***
	Art 2- Control	0.199	215	4.370	>	***
	Art 2- Geotile	0.203	215	5.083	>	***
	Art 2- Grooved	0.199	215	4.168	>	**
	Art 3- Singapore	0.217	215	-5.203	<	***
	Barnacle- Singapore	0.209	215	-6.572	<	***
	Clearing- Singapore	0.208	215	-4.863	<	***
	Control-Singapore	0.208	215	-6.243	<	***
	Geotile- Singapore	0.212	215	-6.897	<	***
	Grooved- Singapore	0.208	215	-6.049	<	***
Saltcoats 12	Art 2- Barnacle	0.238	215	3.589	>	**
	Art 2- Control	0.231	215	3.828	>	**
	Art 2- Geotile	0.251	215	3.918	>	**
	Art 2- Grooved	0.231	215	3.703	>	**
	Art 2- Singapore	0.272	215	-5.329	<	***
	Art 3- Singapore	0.307	215	-6.691	<	***
	Barnacle- Singapore	0.278	215	-8.291	<	***
	Clearing- Singapore	0.272	215	-7.721	<	***
	Control- Singapore	0.272	215	-8.585	<	***
	Geotile- Singapore	0.289	215	-8.415	<	***
	Grooved- Singapore	0.272	215	-8.479	<	***
Saltcoats 18	Art 2- Singapore	0.449	215	-5.492	<	***
	Art 3- Singapore	0.509	215	-5.858	<	***
	Barnacle- Singapore	0.459	215	-7.032	<	***
	Clearing- Singapore	0.449	215	-7.090	<	***
	Control- Singapore	0.449	215	-7.496	<	***
	Geotile- Singapore	0.480	215	-7.091	<	***
	Grooved- Singapore	0.449	215	-7.458	<	***
Site	**Comparison**	**SE**	**df**	**T ratio**	**Direction**	**P value**
IOW 2	Art 3- Clearing	0.138	226	-3.263	<	*
IOW 12	Art 2- Control	0.096	226	3.195	>	*
IOW 18	Art 1- Control	0.134	226	3.973	>	**
	Art 2- Control	0.149	226	3.795	>	**
	Art 3- Control	0.136	226	3.241	>	*

Site	Comparison	SE	df	T ratio	Direction	P value
Blackness 2	Art 1- Clearing	0.212	203	-3.810	<	**
	Art 1- Singapore	0.212	203	-4.612	<	***
	Barnacle- Clearing	0.211	203	-3.819	<	**
	Barnacle- Singapore	0.211	203	-4.624	<	***
	Clearing- Control	0.211	203	4.483	>	***
	Clearing- Grooved	0.211	203	4.885	>	***
	Control- Singapore	0.211	203	-5.287	<	***
	Geotile- Singapore	0.211	203	-3.196	<	*
	Grooved- Singapore	0.211	203	-5.690	<	***
Blackness 6	Art 1- Singapore	0.171	203	-5.466	<	***
	Barnacle- Clearing	0.159	203	-3.788	<	**
	Barnacle- Singapore	0.159	203	-6.879	<	***
	Clearing- Singapore	0.159	203	-3.091	<	*
	Control- Singapore	0.159	203	-6.014	<	***
	Geotile- Singapore	0.159	203	-5.885	<	***
	Grooved- Singapore	0.159	203	-6.043	<	***
Blackness 12	Art 1- Singapore	0.255	203	-3.553	<	**
	Barnacle- Singapore	0.230	203	-5.102	<	***
	Clearing- Singapore	0.230	203	-3.028	<	*
	Control- Singapore	0.230	203	-3.742	<	**
	Geotile- Singapore	0.230	203	-4.805	<	***
	Grooved- Singapore	0.230	203	-3.542	<	**
Blackness 18	Barnacle- Singapore	0.288	203	-4.227	<	***
	Geotile- Singapore	0.288	203	-4.171	<	***

Table A 4-6. Summary of significant post-hoc comparisons for mobile species abundance between tile types by months after installation at each site. GLS models were used for all sites (***=p<0.001, **=p<0.01, *=p<0.05).

Site	Comparison	SE	df	T ratio	Direction	P value
Saltcoats 2	Art 2- Barnacle	0.234	215	3.925	>	**
	Art 2- Control	0.237	215	3.758	>	**
	Art 2- Geotile	0.237	215	3.741	>	**
	Art 2- Grooved	0.240	215	3.621	>	**
	Art 3- Singapore	0.303	215	-3.365	<	*
	Barnacle- Singapore	0.233	215	-5.393	<	***
	Control- Singapore	0.235	215	-5.210	<	***
	Geotile- Singapore	0.236	215	-5.193	<	***
	Grooved- Singapore	0.238	215	-5.054	<	***
Saltcoats 6	Art 2- Art 3	0.223	215	3.156	>	*
	Art 2- Barnacle	0.172	215	6.108	>	***
	Art 2- Control	0.173	215	6.179	>	***

	Art 2- Geotile	0.173	215	6.190	>	***
	Art 2- Grooved	0.175	215	5.546	>	***
	Art 3- Singapore	0.225	215	-4.704	<	***
	Barnacle- Clearing	0.176	215	-3.295	<	*
	Barnacle- Singapore	0.174	215	-8.064	<	***
	Clearing- Control	0.177	215	3.380	>	*
	Clearing- Geotile	0.177	215	3.393	>	*
	Clearing- Singapore	0.245	215	-3.351	<	*
	Control- Singapore	0.175	215	-8.124	<	***
	Geotile- Singapore	0.175	215	-8.134	<	***
	Grooved- Singapore	0.177	215	-7.748	<	***
Saltcoats	Art 2- Barnacle	0.171	215	7.280	>	***
12	Art 2- Control	0.176	215	7.587	>	***
	Art 2- Geotile	0.176	215	7.641	>	***
	Art 2- Grooved	0.183	215	6.139	>	***
	Art 3- Singapore	0.285	215	-3.910	<	**
	Barnacle- Clearing	0.184	215	-3.173	<	*
	Barnacle- Singapore	0.207	215	-7.841	<	***
	Clearing- Control	0.189	215	3.574	>	*
	Clearing- Geotile	0.189	215	3.633	>	**
	Clearing- Singapore	0.263	215	-3.953	<	**
	Control- Singapore	0.211	215	-8.116	<	***
	Geotile- Singapore	0.211	215	-8.162	<	***
	Grooved- Singapore	0.217	215	-6.919	<	***
Saltcoats	Art 2- Barnacle	0.272	215	5.286	>	***
18	Art 2- Control	0.284	215	5.652	>	***
	Art 2- Geotile	0.284	215	5.719	>	***
	Art 2- Grooved	0.297	215	4.305	>	***
	Barnacle- Singapore	0.337	215	-5.461	<	***
	Control- Singapore	0.346	215	-5.788	<	***
	Geotile- Singapore	0.347	215	-5.844	<	***
	Grooved- Singapore	0.357	215	-4.701	<	***
Site	**Comparison**	**SE**	**df**	**T ratio**	**Direction**	**P value**
IOW	Art 1- Clearing	0.202	226	-7.135	<	***
2	Art 2- Clearing	0.115	226	-15.005	<	***
	Art 3- Clearing	0.112	226	-16.366	<	***
	Barnacle- Clearing	0.075	226	-25.000	<	***
	Clearing- Control	0.159	226	10.996	>	***
	Clearing- Geotile	0.092	226	19.659	>	***
	Clearing- Grooved	0.174	226	9.923	>	***
IOW	Art 1- Clearing	0.154	226	-6.242	<	***
6	Art 1- Control	0.180	226	3.518	>	*
	Art 2- Clearing	0.120	226	-9.149	<	***
	Art 2- Control	0.153	226	3.215	>	*
	Art 3- Clearing	0.095	226	-13.067	<	***
	Barnacle- Clearing	0.088	226	-15.555	<	***
	Clearing- Control	0.122	226	13.066	>	***
	Clearing- Geotile	0.094	226	13.847	>	***
	Clearing- Grooved	0.136	226	9.048	>	***

IOW	Art 1- Control	0.217	226	5.203	>	***
12	Art 2- Control	0.254	226	4.688	>	***
	Art 3- Control	0.150	226	6.733	>	***
	Barnacle- Clearing	0.193	226	-3.184	<	*
	Barnacle- Control	0.187	226	4.004	>	**
	Clearing- Control	0.155	226	8.809	>	***
	Control- Geotile	0.191	226	-4.221	<	***
	Control- Grooved	0.180	226	-4.945	<	***
IOW	Art 1- Control	0.362	226	4.497	>	***
18	Art 2- Control	0.416	226	4.547	>	***
	Art 3- Control	0.244	226	6.831	>	***
	Barnacle- Control	0.302	226	4.235	>	***
	Clearing- Control	0.250	226	4.535	>	***
	Control- Geotile	0.186	226	-4.399	<	***
	Control- Grooved	0.180	226	-4.930	<	***
Site	**Comparison**	**SE**	**df**	**T ratio**	**Direction**	**P value**
Blackness	Art 1- Clearing	0.279	203	-5.340	<	***
2	Art 1- Singapore	0.175	203	-10.518	<	***
	Barnacle- Clearing	0.251	203	-6.511	<	***
	Barnacle- Singapore	0.125	203	-15.884	<	***
	Clearing- Control	0.294	203	5.243	>	***
	Clearing- Geotile	0.251	203	6.500	>	***
	Clearing- Grooved	0.251	203	6.494	>	***
	Control- Singapore	0.199	203	-9.516	<	***
	Geotile- Singapore	0.125	203	-15.822	<	***
	Grooved- Singapore	0.126	203	-15.780	<	***
Blackness	Art 1- Clearing	0.197	203	-4.796	<	***
6	Art 1- Singapore	0.166	203	-7.238	<	***
	Barnacle- Clearing	0.166	203	-7.457	<	***
	Barnacle- Grooved	0.172	203	-4.288	<	***
	Barnacle- Singapore	0.126	203	-11.849	<	***
	Clearing- Control	0.184	203	5.116	>	***
	Clearing- Geotile	0.190	203	5.484	>	***
	Control- Singapore	0.151	203	-7.916	<	***
	Geotile- Singapore	0.156	203	-8.315	<	***
	Grooved- Singapore	0.187	203	-4.041	<	**
Blackness	Art 1- Singapore	0.264	203	-3.034	<	*
12	Barnacle- Clearing	0.227	203	-4.331	<	***
	Barnacle- Grooved	0.280	203	-4.280	<	***
	Barnacle- Singapore	0.199	203	-5.919	<	***
	Control- Singapore	0.225	203	-3.373	<	*
	Geotile- Singapore	0.250	203	-3.455	<	*
Blackness	Barnacle- Grooved	0.344	203	-4.275	<	***
18	Barnacle- Singapore	0.249	203	-3.998	<	**

Chapter 5 Appendix

Table A 5-1. References used in systematic review in Chapter 5.

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5.1. August 2016 surveys

5.1.1. Method

In August 2016, seven habitats were surveyed (n=3 natural and n=4 artificial), with some habitat types sampled along the length of the scheme (Table A 5.1-1). Five 25x25 cm quadrats were randomly placed in each plot, apart from the rock armour sampling where 4 quadrats were sampled in one plot due to tidal constraints.

Table A 5.1-1. Number of quadrats and number of plots for August 2016 surveys.

Sampling	Quadrat Numbers	Number of survey plots
Concrete Apron Horizontal	25	5
Platform Horizontal	15	3
Concrete Apron Vertical	20	4
Platform Vertical	5	1
Rock Armour	9	2
Concrete Wall Vertical	20	4
Platform Horizontal Abraded	5	1

5.1.2. Results

Comparing the grouped artificial habitat types against the grouped natural habitat types revealed that natural shore platform habitats have greater species richness than artificial habitats ($H[1]=12.408$, $p=0.00043$). No differences were evident between natural and artificial habitats for mobile species abundance ($H[1]=0.8632$, $p=0.3528$) and algae and lichen species abundance ($H[1]= 0.71215$, $p=0.3987$).

In examining individual habitat types, abbreviations for habitats are as follows: concrete apron horizontal (CAH), platform horizontal (PH), concrete apron vertical (CAV), platform vertical (PV), rock armour (RA), concrete wall vertical (CWV), platform horizontal abraded (PHA). Differences between habitat types influenced species richness ($H[6]= 29.928$, $p = 4.057e-05$), mobile species abundance ($H[6]=28.443$, $p=7.753e-05$) and algae and lichen species abundance ($H[6]=27.141$, $p=0.00014$) (Figure A 5.1-1). These analyses excluded *Semibalanus balanoides, Actinia equina* and *Polydora ciliata.* All three of these species are examined individually in the next section.

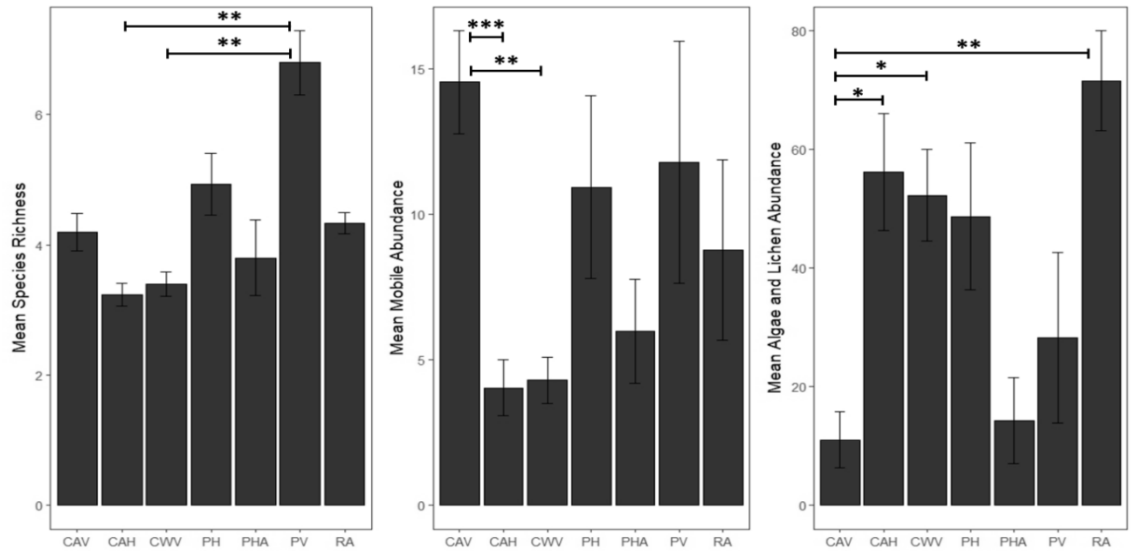

Figure A 5.1-1. Mean species richness and abundance across all habitat types at Hartlepool. Dunn's post-hoc results for significance are reported with Bonferroni adjusted p-values (0.05/21= (0.0024); *=P<0.0024, **=p<0.001, ***=p<0.0001 ($\bar{x}$ ± standard error).

Vertical natural shore platforms had greater species richness than surrounding artificial habitats (Figure A 5.1-1). Species richness was significantly greater on the natural vertical platform (PV) (6.8±0.49 SE), than on the vertical concrete wall (3.4±0.18 SE) and horizontal concrete apron (3.24±0.18 SE) (*z*=-3.95, p=0.0008 and z=-4.18, p=0.0003 respectively) (Figure A 5.1-1). In contrast, species abundance tended to be higher on artificial than natural habitats. Mobile species abundance was greater on the CAV than CAH (z=-4.816051, p=0.0000) and the CWV (z=4.083523, p= 0.0005). At this site, *Patella vulgata* was the dominant mobile species and was commonly observed across all habitat types, in particular on the CAV. Algae and lichen abundance was greater on the CAH (z=3.752708, p=0.0018), the CWV (z=-3.905106, p=0.0010) and RA (-4.080381, p=0.0005) than the CAV. Although not significant, the abraded horizontal platform (PHA) can be seen to have the lowest species richness and abundance of the natural habitats examined (Figure A 5.1-1).

The most common species across both natural and artificial habitats were *Littorina littorea, Patella vulgata* (Figure A 5.1-2)*, Semibalanus balanoides, Fucus vesiculosus, Ulva* (spp.) and *Rhodothamniella floridula* (Figure A 5.1-3). Two varieties of *Ulva* (spp.) were sampled but they were not identified to species level as identification within this genus is difficult and would require destructive sampling and identification at a cellular level. The same applies to *Verrucaria* (sp.).

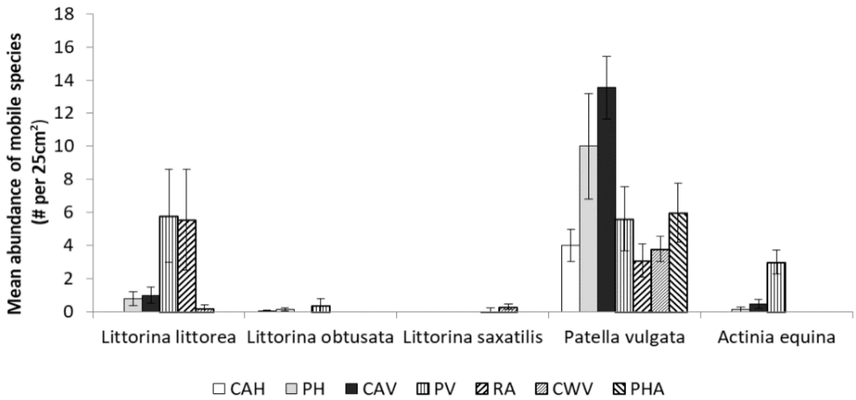

Figure A 5.1-2. Mean mobile species abundance on different habitat types along the Hartlepool headland defence scheme. Numbers of quadrats for each habitat are noted in brackets: CAH (n=25), PH (n=15), CAV (n=20), PV (n=5), RA (n=9), CWV (n=20), PHA (n=5) ($\bar{x}$ ± standard error).

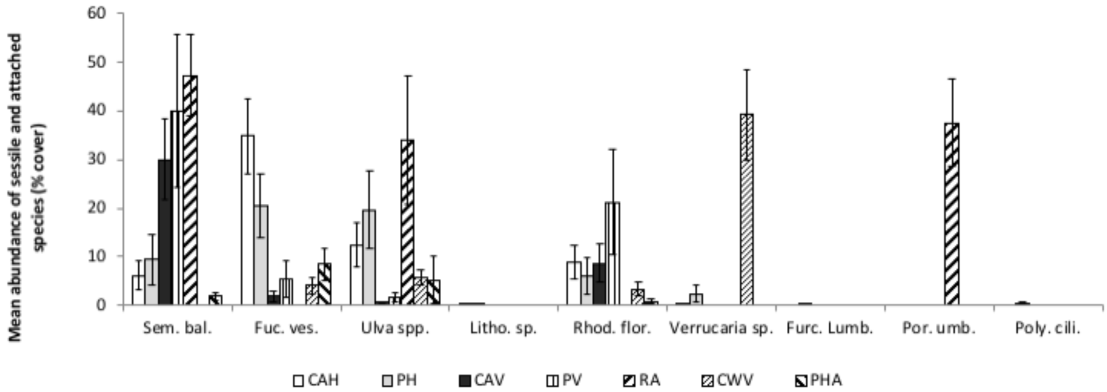

Figure A 5.1-3. Mean sessile and attached (algae and lichen) abundance (%) on different habitat types along the Hartlepool headland defence scheme. CAH (n=25), PH (n=15), CAV (n=20), PV (n=5), RA (n=9), CWV (n=20), PHA (n=5). ($\bar{x}$ ± standard error).

A total of 14 taxa were recorded across all habitats surveyed, with an average of 7 species recorded in artificial habitats and 10 species across the natural horizontal and vertical shore platforms. Even when including abraded areas, notoriously species-poor (Table A 5.1-2), the average number of species recorded for natural habitats (8.33) was still higher than artificial habitats. PH habitat had the greatest number of species recorded, even if on average the PV habitat had a greater species richness, as reported earlier. Abraded habitat, that had been created during the construction of the defences, had fewer species present than artificial habitats (Table A 5.1-2). Rock armour surveyed was also notably species-poor at this stage of construction, so it is hoped that passive enhancement recommendations will improve the number of species present on this structure. Differences in numbers of species present were not statistically significant between individual habitats (*H*[6] =6, p=0.4232).

Table A 5.1-2. Species presence in natural (white) and artificial (grey) habitats.

Species	CAH	PH	CAV	PV	RA	PHA	CWV
Littorina littorea		✓	✓	✓	✓		✓
Littorina obtusata	✓	✓		✓			
Patella vulgata	✓	✓	✓	✓	✓	✓	✓
Semibalanus balanoides	✓	✓	✓	✓	✓	✓	
Fucus vesiculosus	✓	✓	✓	✓		✓	✓
Ulva (spp.)	✓	✓	✓	✓	✓	✓	✓
Lithothamnion (sp.)	✓	✓					
Actinia equina		✓	✓	✓			
Rhodothamniella floridula	✓	✓	✓	✓		✓	✓
Verrucaria sp.	✓	✓					✓
Littorina saxatilis					✓		✓
Polydora ciliata		✓					
Furcellaria lumbricalis		✓					
Porphyra umbilicalis					✓		
Total Species Recorded	8	12	7	8	6	5	7

In examining the distributions of individual species between natural and artificial habitats, habitat preferences differ for each species. *L. obtusata* was observed on PV (n=2), PH (n=2) and CAH (n=1) and did not differ between habitat types due to their occurrence in very low abundances. The same can be said of *L. saxatilis,* which was unique to artificial habitats, with observations on rock armour (n=1) and CWV (n=6) in low numbers that did not result in any significant variation in habitat distribution.

The vertical platform proved to be important habitat for both *L .littorea* and in particular *A.equina,* which is typically found attached to hard substrata. *A.equina* was observed in higher abundance on the PV (3±0.71 SE) on average than on all other habitats (p<0.0001) (Table A 5.1-3). *L. littorea* also frequented PV habitat (5.8± 2.84 SE), with greater numbers observed than on both the CAH, CWV (p<0.0001 for both) and the PHA (p<0.001).

Polydora ciliata were unique to the Magnesian limestone horizontal shore platform as they burrow into limestone rock and extend above the substratum surface. They occurred in low abundances (0.4% ± 0.18 SE) but were found in significantly greater abundances than all examined artificial habitats (p<0.001) (Table A 5.1-3).

S. balanoides were observed across all habitat types except the vertical concrete wall, which was too high in the tidal zone. Similar mean percentage covers of barnacles were observed on the natural vertical platform (40% ± 15.65 SE) compared to the newly installed rock armour (47.28%± 8.30 SE). The vertical concrete apron was also covered in a relatively high cover of barnacles compared to other artificial habitats (30%± 8.37 SE). All three of these habitats had a greater abundance of barnacles than the CWV, likely due to its position in the tidal frame (p<0.001). The CAV and the RA also had greater barnacle cover than the horizontal concrete apron (p<0.0024).

Limited differences were detected between habitat types for algae species. *F. vesiculosus* was more abundant on the CAH than on rock armour (z=3.80815, p=0.0015). *P. umbilicalis* was unique to rock armour and so had significantly greater cover (37.56% ± 8.89 SE) than all other habitats (p<0.0001). *F. lumbricalis* was unique to the PH but occurred in such a low abundance (0.07% ± 0.07 SE) that there was no notable difference with other habitats. It should be noted that whilst there was no significant differences in the early successional *Ulva* (spp.), more was observed on the rock armour than in other habitats (34.06% ± 13.22 SE), indicating it is in an earlier stage of succession than adjacent platform habitats. *Verrucaria* (sp.) was also frequently observed on the CWV compared to CAH, CAV, PH and RA habitats (p<0.001). Further statistical information can be found in Table A 5.1-3.

Table A 5.1-3. Summary of the results of Kruskal-Wallis testing and Dunn's post-hoc with Bonferroni adjusted p-values for multiple comparisons of mobile and sessile species between habitat types (0.05/21= (0.0024); *=P<0.0024, **=p<0.001, ***=p<0.0001).

Class	Species	H	df	p	Dunn's test	
Gastropoda	*Littorina littorea*	34.407	6	5.612e-06	PV > CAH	***
					PV > CWV	***
					PV > PHA	**
	Littorina obtusata	8.9636	6	0.1756	NS	
	Littorina saxatilis	10.237	6	0.115	NS	

	Patella vulgata	25.374	6	0.0002911	CAV > CAH CAV > CWV	** **
Anthozoa	*Actinia equina*	56.59	6	2.211e-10	PV > ALL	***
Polychaeta	*Polydora ciliata*	35.376	6	3.644e-06	PH > CAH PH > CAV PH > CWV PH > RA	*** *** *** **
Maxillopoda	*Semibalanus balanoides*	60.932	6	2.91e-11	CAV > CAH CAV > CWV PV > CWV RA > CAH RA > CWV	** *** ** * ***
Phaeophyceae	*Fucus vesiculosus*	24.568	6	0.0004104	CAH > RA	*
Ulvophyceae	*Ulva* (spp.)	13.876	6	0.03105	NS	
Bangiophyceae	*Porphyra umbilicalis*	97.717	6	< 2.2e-16	RA > ALL	***
Florideophyceae	*Rhodothamniella floridula*	11.409	6	0.07652	NS	
	Lithothamnion (sp.)	7.1413	6	0.308	NS	
	Furcellaria lumbricalis	5.6	6	0.4695	NS	
Eurotiomycetes	*Verrucaria sp.*	49.675	6	5.461e-09	CWV > CAH CWV > CAV CWV > PH CWV > RA	*** *** ** **

In order to compare the natural and artificial habitats in more depth within this site, a series of habitat comparisons were undertaken examining similar habitat types, with vertical habitats and horizontal habitats compared (Table A 5.1-4). Full statistical results from tests on species abundances across habitat comparisons can be seen in Table A 5-A 7.

For key intertidal prey species for waterbirds (*Littorina littorea, Littorina obtusata, Patella vulgata*), vertical habitat is important. In particular with *P. vulgata,* numbers were significantly higher on the concrete apron than rock armour quadrats as the concrete apron offers a ledge-like habitat, with limpets concentrated at the base of the vertical apron. For *Littorinidae,* the vertical platform showed to be optimal habitat compared to concrete vertical habitat (Table A 5.1-4). Early successional species of *Ulva* (spp.) were more common on rock armour than the concrete apron ($p<0.001$) that had a longer exposure time to the intertidal environment. Most of the results in comparing horizontal habitat types were not significant. Although previous tests in this chapter have shown the importance of the horizontal platform, there are limited differences between the PH and CAH (Table A 5.1-4). *Polydora ciliata* were unique to the platform as they require substrates they can burrow into and so had a higher abundance than the concrete apron ($p<0.001$). No significant differences were found between the horizontal platform and the abraded platform.

Table A 5.1-4. Summary of the results of Kruskal-Wallis testing (for two way comparisons) and Dunn's post-hoc testing (for three way comparison) of species abundance in habitat comparisons of natural and artificial, vertical and horizontal habitats. Bonferroni adjusted p-values were used for multiple comparisons (PV vs CAV vs RA- 0.05/3= (0.016); *=p<0.016, **=p<0.01, ***=p<0.001) otherwise standard significance measures were used (*=p<0.05, **=p<0.01, ***=p<0.001). NS = Not significant, X= species absent from habitats.

	Vertical Habitat		**Horizontal Habitat**	
Species Abundance	**PV vs CAV vs RA**	**CWV vs PV**	**CAH vs PH**	**PH vs PHA**
Littorina littorea	[PV > CAV] **	[PV > CWV] ***	[PH>CAH] **	NS
Littorina obtusata	NS	[PV > CWV] *	NS	NS
Littorina saxatilis	NS	NS	X	X
Patella vulgata	[CAV > RA] ***	NS	NS	NS
Actinia equina	[PV> both] ***	[PV > CWV] ***	NS	NS
Polydora ciliata	X	X	[PH>CAH] ***	NS
Semibalanus balanoides	NS	[PV > CWV] ***	NS	NS
Fucus vesiculosus	[PV > RA] **	NS	NS	NS
Ulva (spp.)	[RA > CAV] ***	NS	NS	NS

Porphyra umbilicalis	[RA >both] ***	X	X	X
Rhodothamniella floridula	[PV > RA] **	[CWV > PV] *	NS	NS
Lithothamnion (sp.)	X	X	NS	NS
Furcellaria lumbricalis	X	X	NS	NS
Verrucaria sp.	X	[CWV > PV] *	NS	NS

Table A 5.1-5. Summary of the results of Kruskal-Wallis testing of August 2016 habitat comparisons of mobile, sessile and attached species on the concrete apron horizontal (CAH) and platform horizontal (PH) ($p<0.05$*, $p<0.01$**, $p<0.001$***).

CAH vs PH	**H**	**df**	**p**	**Comparison**
Littorina littorea	7.2072	1	**	PH >CAH
Littorina obtusata	1.1477	1	NS	
Patella vulgata	2.6130	1	NS	
Semibalanus balanoides	2.9023	1	NS	
Fucus vesiculosus	0.5264	1	NS	
Ulva (spp.)	0.03285	1	NS	
Lithothamnion (sp.)	1.0189	1	NS	
Actinia equina	1.6667	1	NS	
Rhodothamniella floridula	0.01457	1	NS	
Verrucaria sp.	2.6781	1	NS	
Littorina saxatilis	NA			
Polydora ciliata	11.4040	1	***	PH > CAH
Furcellaria lumbricalis	1.6667	1	NS	
Porphyra umbilicalis	NA			

Table A 5.1-6. Summary of the results of Kruskal-Wallis testing and Dunn's post-hoc testing for August 2016 habitat comparisons of mobile, sessile and attached species on the platform vertical (PV), concrete apron vertical (CAV) and rock armour (RA). Bonferroni adjusted p-values were used for multiple comparisons (0.05/3= (0.016); *=p<0.016, **=p<0.01, ***=p<0.001).

PV vs CAV vs RA	H	df	p	Comparison	Dunn's test	p	
Littorina littorea	7.7765	2	0.02048	CAV < PV	-2.7890	0.0079	**
Littorina obtusata	5.8	2	0.05502		NS		
Patella vulgata	13.9530	2	0.00093	CAV > RA	3.6060	0.0005	***
Semibalanus balanoides	1.7026	2	0.4269		NS		
Fucus vesiculosus	8.8736	2	0.0118	PV > RA	2.9750	0.0044	**
Ulva (spp.)	12.738	2	0.0017	CAV < RA	-3.5596	0.0006	***
Actinia equina	18.8240	2	8.173e-05	CAV < PV	-3.9050	0.0001	***
				PV > RA	4.1182	0.0001	***
Rhodothamniella floridula	8.8612	2	0.01191	PV > RA	2.8434	0.0067	**
Littorina saxatilis	2.7778	2	0.2494		NS		
Porphyra umbilicalis	32.029	2	1.109e-07	CAV < RA	-5.4809	0.0000	***
				PV < RA	-3.9442	0.0001	***

Table A 5.1-7. Summary of the results of Kruskal-Wallis testing of August 2016 habitat comparisons of mobile, sessile and attached species on the vertical platform (PV) and vertical concrete wall (CWV) (p<0.05*, p<0.01**, p<0.001***).

PV vs CWV	H	df	p	Comparison
Littorina littorea	18.2190	1	***	CWV < PV
Littorina obtusata	4	1	*	CWV < PV
Patella vulgata	0.9910	1	NS	
Semibalanus balanoides	23.6220	1	***	CWV < PV
Fucus vesiculosus	0.5462	1	NS	
Ulva (spp.)	0.6407	1	NS	
Actinia equina	23.6220	1	***	CWV < PV
Rhodothamniella floridula	5.2617	1	*	CWV > PV
Verrucaria sp.	6.2104	1	*	CWV > PV
Littorina saxatilis	0.8142	1	NS	

Chapter 6 Appendix

Table A 6-1. Summary of GLM results for pits from natural shore surveys for species richness and mobile abundance for width, depth and water holding capacity. Significant dimensions from post-hoc testing are also reported (***=p<0.001, **=p<0.01, *=p<0.05).

a) Pits species richness- Quasi-Poisson				
	Estimate	**Std. Error**	**T value**	**P**
(Intercept)	-0.139	1.000	-0.139	NS
Width	0.036	0.026	1.393	NS
Depth	-0.026	0.021	-1.226	NS
Water holding %	0.018	0.026	0.681	NS
b) Pits mobile abundance- Negative Binomial				
	Estimate	**Std. Error**	**Z value**	**P**
(Intercept)	3.620	0.918	3.943	***
Width	-0.042	0.027	-1.552	NS
Depth	-0.010	0.018	-0.565	NS
Water holding %	0.072	0.026	2.770	**
c) Pits mobile abundance- Quasi-Poisson post-hoc significant dimensions				
Width (mm)	**Estimate**	**Std. Error**	**Z value**	**P**
30-40 – 20-30 == 0	2.818	0.557	5.059	***
40-50 – 20-30 == 0	1.729	0.456	3.796	***
40-50 – 30-40 == 0	-1.088	0.320	-3.397	**
Depth (mm)				
20-30 – 10-20 == 0	-1.617	0.361	-4.480	***
30-40 – 10-20 == 0	-1.722	0.434	-3.967	***
50-60 – 10-20 == 0	-2.335	0.785	-2.976	*
Water holding in quadrat %				
10-20 – 0-10 == 0	-1.814	0.462	-3.930	***

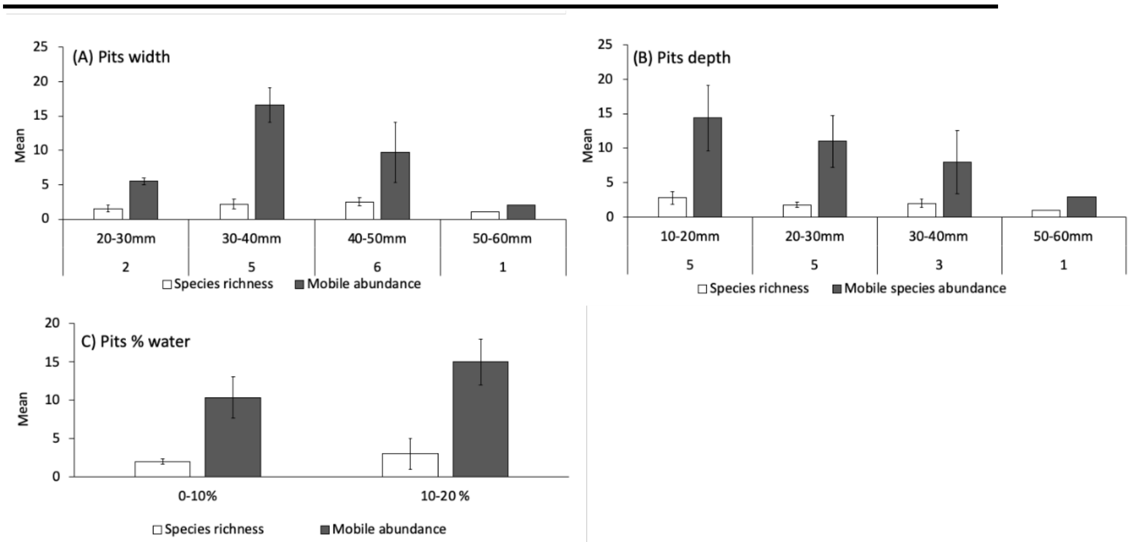

Figure A 6-1. Mean species richness and mobile species abundance for varying dimensions of width and depth and % water in pits (n=14) (x̅ ± standard error).

Table A 6-2. Summary of GLM results for deep pools from natural shore surveys for species richness, mobile abundance and algae and lichen abundance for width, depth and water holding capacity. Significant dimensions from post-hoc testing are also reported (***=p<0.001, **=p<0.01, *=p<0.05).

a) Deep pool species richness- Quasi-Poisson				
	Estimate	**Std. Error**	**T value**	**P**
(Intercept)	1.266	0.151	8.399	***
Width	-0.001	0.000	-1.147	NS
Depth	0.004	0.002	2.441	*
b) Deep pool mobile abundance- Negative Binomial				
	Estimate	**Std. Error**	**Z value**	**P**
(Intercept)	1.758	0.361	4.869	***
Width	0.001	0.001	0.517	NS
Depth	0.005	0.004	1.031	NS
c) Algae and lichen abundance- Linear model				
	Estimate	**Std. Error**	**T value**	**P**
(Intercept)	13.483	8.810	1.531	NS
Width	-0.009	0.025	-0.345	NS
Depth	0.278	0.110	2.527	*

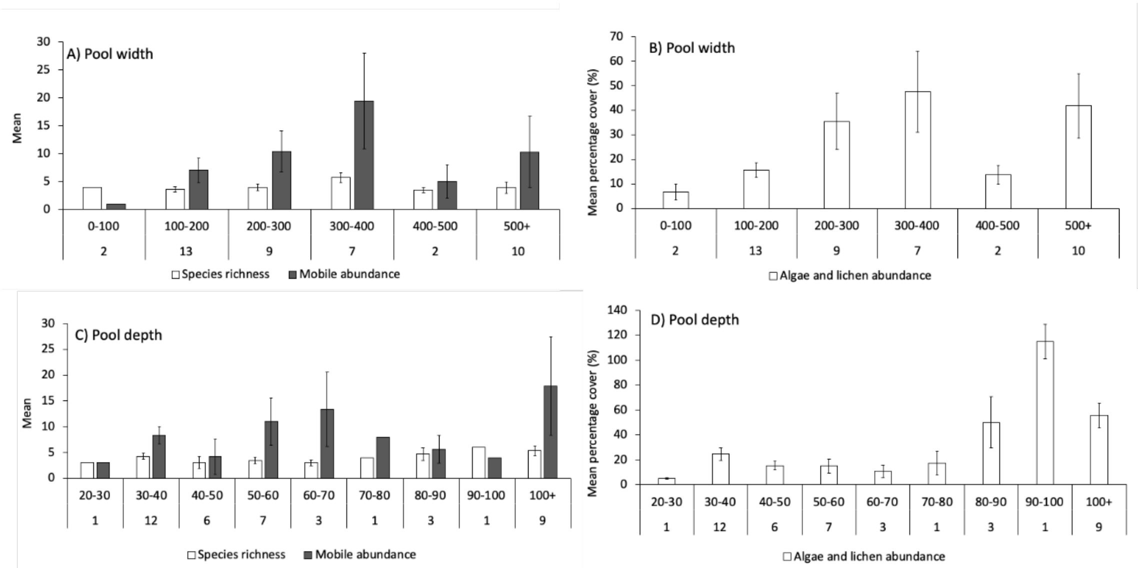

Figure A 6-2. Mean species richness, mobile species abundance and algae and lichen abundance (%) for varying dimensions of width and depth of deep pools(n=43) ($\bar{x}$ ± standard error).

Table A 6-3. Summary of GLM results for crevices from natural shore surveys for species richness and mobile abundance for width, depth and water holding capacity. Significant dimensions from post-hoc testing are also reported (***=p<0.001, **=p<0.01, *=p<0.05).

a) Crevice species richness- Quasi-Poisson				
	Estimate	**Std. Error**	**T value**	**P**
(Intercept)	0.698	0.165	4.220	***
Width	-0.007	0.005	-1.490	NS
Depth	0.006	0.003	1.986	NS
Water holding %	0.015	0.006	2.396	*
b) Crevice species richness- Quasi-Poisson post-hoc significant dimensions				
Depth (mm)	**Estimate**	**Std. Error**	**Z value**	**P**
90-100 – 50-60 == 0	1.696	0.505	3.358	*
c) Crevice mobile abundance- Negative Binomial				
	Estimate	**Std. Error**	**Z value**	**P**
(Intercept)	1.281	0.213	6.004	***
Width	-0.004	0.006	-0.582	NS
Depth	0.013	0.004	2.985	**
Water holding %	-0.003	0.009	-0.294	NS
d) Crevices mobile abundance- Negative Binomial post-hoc significant dimensions				
Depth (mm)	**Estimate**	**Std. Error**	**Z value**	**P**
90-100 – 30-40 == 0	1.917	0.606	0.035	*
e) Algae and lichen abundance- Linear model				
	Estimate	**Std. Error**	**T value**	**P**
(Intercept)	2.213	1.462	1.514	NS
Width	-0.030	0.042	-0.724	NS
Depth	0.008	0.030	0.251	NS
Water holding %	0.166	0.061	2.734	**
f) Algae and lichen abundance- Post-hoc significant dimensions				
Depth (mm)	**Estimate**	**Std. Error**	**T value**	**P**
90-100 – 20-30 == 0	17.158	4.612	3.720	*
90-100 – 30-40 == 0	14.809	4.308	3.438	*
90-100 – 50-60 ==0	15.772	4.357	3.620	*
90-100 – 70-80 ==0	14.646	4.353	3.364	*

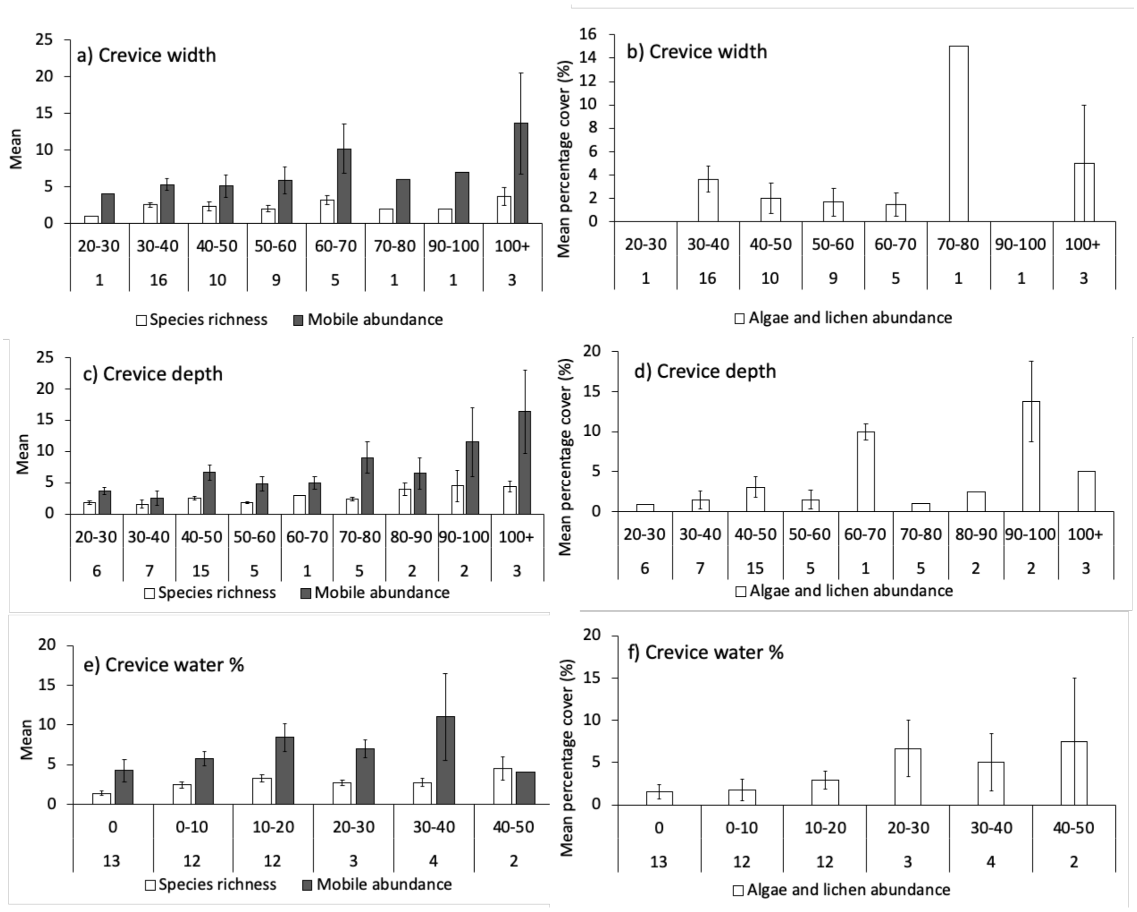

Figure A 6-3. Mean species richness, mobile species abundance and algae and lichen abundance (%) for varying dimensions of width and depth of crevices (n=46) ($\bar{x}$ ± standard error).

Table A 6-4. Summary of GLM results for ledges from natural shore surveys for species richness and mobile abundance for width, height and water holding capacity. Significant dimensions from post-hoc testing are also reported (***=p<0.001, **=p<0.01, *=p<0.05).

a) Ledge species richness- Quasi-Poisson				
	Estimate	**Std. Error**	**T value**	**P**
(Intercept)	0.467	0.124	3.772	***
Width	-0.001	0.000	-3.040	**
Height	0.003	0.000	5.525	***
b) Ledge species richness- Quasi-Poisson post-hoc significant dimensions				
Width (mm)	**Estimate**	**Std. Error**	**T value**	**P**
500-600 – 100-200 == 0	-1.000	0.333	-3.004	*
Height (mm)				
500-600 – 0-50 == 0	1.514	0.431	3.591	*
50-100 – 350-400 == 0	-1.268	0.322	-3.939	**
500-600 – 50-100 == 0	1.473	0.335	4.398	***
c) Ledge mobile abundance- Negative Binomial				

	Estimate	Std. Error	Z value	P
(Intercept)	1.048	0.228	4.591	***
Width	-0.000	0.001	-0.499	NS
Height	0.005	0.001	4.439	***

d) Ledge mobile abundance- Negative Binomial post-hoc significant dimensions

Height (mm)	Estimate	Std. Error	Z value	P
200-250 – 0-50 == 0	2.302	0.676	3.404	*
500-600 – 0-50 == 0	3.417	0.806	4.239	**
600+ – 0-50 == 0	3.406	0.960	3.547	*
500-600 – 50-100 == 0	2.260	0.657	3.439	*

e) Ledge algae and lichen abundance- Linear model

	Estimate	Std. Error	T value	P
(Intercept)	-1.776	2.330	-0.762	NS
Width	0.004	0.006	0.699	NS
Height	0.019	0.011	1.796	NS

f) Ledge algae and lichen abundance- Post-hoc significant dimensions

Width (mm)	Estimate	Std. Error	Z value	P
600+ – 100-200 == 0	14.607	4.817	3.033	*
600+ – 200-300 == 0	15.134	4.546	3.329	*
600+ – 300-400 == 0	15.870	5.071	3.129	*
600+ – 400-500 == 0	17.343	5.527	3.138	*
600+ – 500-600 == 0	19.556	5.449	3.589	*
Height (mm)				
500-600 – 100-150 == 0	23.790	6.832	3.482	*
500-600 – 150-200 == 0	27.580	7.005	3.937	**
500-600 – 300-350 == 0	38.970	9.767	3.989	**
500-600 – 400-450 == 0	30.340	8.680	3.496	*
500-600 – 50-100 == 0	22.260	6.550	3.398	*
600+ – 500-600 == 0	-38.970	9.767	-3.989	**

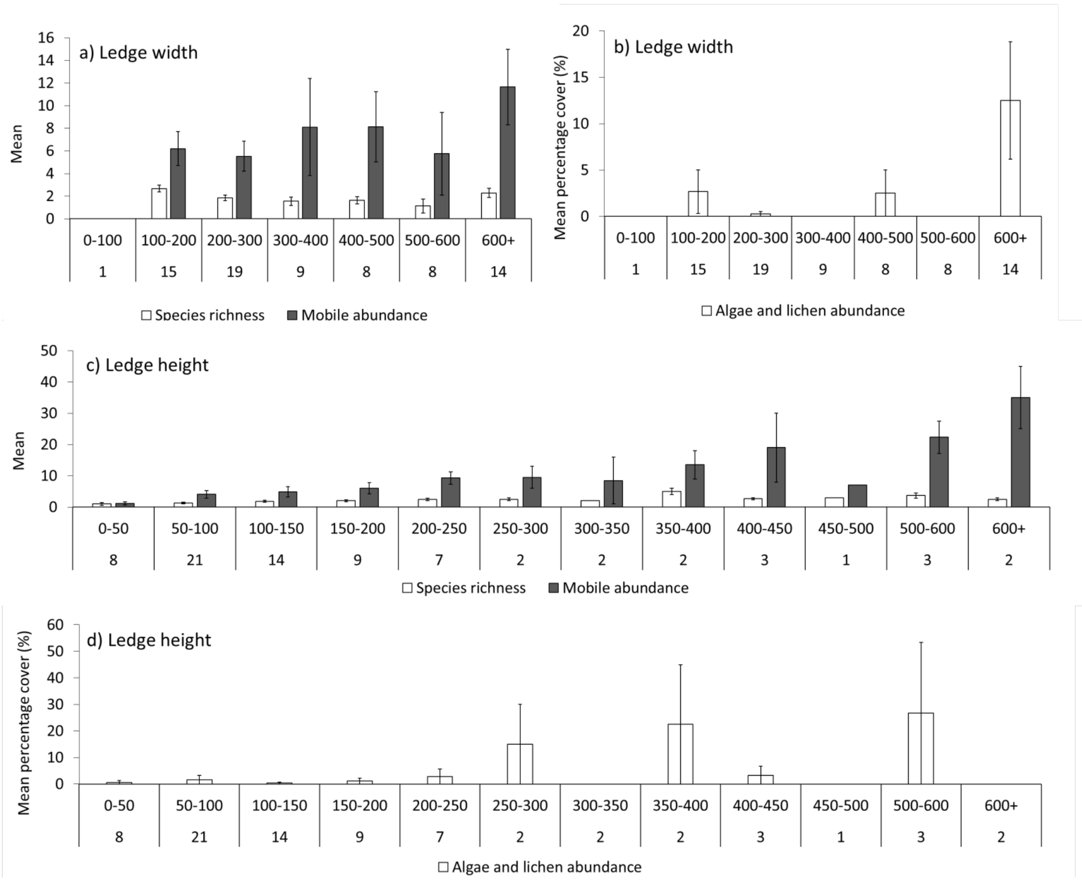

Figure A 6-4. Mean species richness, mobile species abundance and algae and lichen abundance (%) for varying dimensions of width and height of ledges (n=74) (x̅ ± standard error).