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Intertidal Structures: Coastal Engineering for Sustainability and Biodiversity

by

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ABSTRACT

Coastal defence structures (CDS) are propagating globally. Growing environmental concerns are driving modifications to traditional coastal engineering methods through environmentally-enhancing designs which promote biodiversity and socio-economic benefits. Despite a number of tested ecologically-sensitive designs, there is still a gap between research concepts and practical implementations to coastal engineering designs. I investigated design methods for CDS to improve their role as surrogate habitats for coastal assemblages, thus creating more sustainable coastal protection for engineering and maintaining biodiversity. I focused on four key knowledge gaps to identify novel, sustainable and, more importantly, practical methods of designing ecologically sensitive coastal protection: (1) the extent of biological and topographic dissimilarity between natural and artificial shores at different scale levels; (2) the use of porous CDS as multifunctional designs for coastal engineering; (3) the use of 3D printing in coastal engineering to design complexity into defence structures; and finally, (4) the impacts of intertidal and subtidal species and their role as natural coastal protection methods. To address the first knowledge gap, I surveyed seven natural and artificial shores on the South coast, UK, comparing the biological communities and topographic complexity on each shore at three different scale levels. I found that species characteristic of natural and artificial shores differ, and natural shores tend to be characterised by species such as fucoids and some foliose red algae, while artificial shores are largely characterised by invertebrate species. For the second knowledge gap, I surveyed a porous CDS during a groyne reduction process, and compared the coastal assemblages colonising the internal and external habitats of the structure. The results showed significant differences in species richness and diversity on internal habitats to external. For the third knowledge gap, I explored the use of 3D printing to design-in complex habitat features to enhance biodiversity on artificial structures. This study showed colonisation of some coastal species, but more importantly identified key limitations when using this novel material in coastal engineering, which are fundamental at this preliminary stage. Finally, to address the last knowledge gap, I investigated the impact of eight intertidal and subtidal mimic species on wave velocity. The results showed significant reductions in wave velocity due to the presence of all mimics, particularly longer and more flexible species. Additionally, I compared the impact of five different designed tile units on wave velocity. I found significant differences in wave velocity reduction among all tile designs, particularly between units of varying orientations. To conclude my thesis, I summarise the key findings and evaluate these outcomes in the context of their application to sustainable coastal engineering. I then outline challenges and practical methods for designing sustainable and multifunctional coastal defence schemes.

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Glossary

Ecological Engineering	A concept which integrates ecological, economic and societal needs into the design of man-made ecosystems.
Ecoystem Engineers	Organisms that modify, maintain or create habitats through their ability to self-design and/ or self-organise.
Spatial scale	A measurement of size and space in the context of engineering structures (whether micrometres, millimetres, centimetres etc).
Structural Complexity	The physical architecture of a structure which considers the surface roughness and relative abundance of different features such as crevices, holes, overhangs.
Sustainability	The environment, social and economic developments of coastal areas with the goals to invest in the conservation, management and restoration of critical ecosystem functions. Within the context of this thesis, the use of the word *sustainable* will refer to cases where engineering intervention is taking place and therefore considers the protective options available.

Declaration of Authorship

I, Talia Rose Wilson Sherrard, declare that the thesis entitled *Intertidal Structures: Coastal Engineering for Sustainability and Biodiversity* and the work presented in the thesis are both my own, and have been generated by me as the result of my own original research. I confirm that:

- this work was done wholly or mainly while in candidature for a research degree at this University;
- Where any part of this thesis has previously been submitted for a degree or any other qualification at this University or any other institution, this has been clearly stated;
- Where I have consulted the published work of others, this is always clearly attributed;
- Where I have quoted from the work of others, the source is always given. With the exception of such quotations, this thesis is entirely my own work;
- I have acknowledged all main sources of help;
- Where the thesis is based on work done by myself jointly with others, I have made clear exactly what was done by others and what I have contributed myself;
- Parts of this work have been published as: Sherrard et al. (2015) and Sherrard et al. (2016).

Signed:...

Date:...

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Chapter 1

Introduction and Thesis Aims

1.1 Introduction

Coastal zones occupy less than 15% of the Earth's land surface cover (EEA, 2006) and yet 37% of the global population live within close proximity of coastal areas (Cohen et al., 1997; Small and Nicholls, 2003; Nicholls et al., 2015). This is adding intense pressures on our coastal resources and assets (Airoldi and Beck, 2007; McGranahan et al., 2007; Bulleri and Chapman, 2010). Combined with the impacts of climate change, such as sea level rise (IPCC, 2007a; Nicholls et al., 2014, 2015) and stormier seas (Marshall, 2001), and an increase in urban development, there is a growing demand for coastal defence structures (CDSs). Artificial structures date back as far as sedentary man where they were used to hold back the advances of the sea (Charlier et al., 2005), reduce erosion and support navigation needs. Since then, CDSs have continued to develop forming various types of structural designs in order to recover, dam, drain and transform land areas susceptible to the effects of nearby water bodies (Charlier et al., 2005). Changes in sea-level and shoreline erosion have been a consistent pressure on coastal areas, however, increased urbanisation adds growing drivers for further CDSs in order to protect coastal assets.

To continue protecting our natural and developed coasts, engineers and environmental practitioners need to understand the impacts of artificial structures and their roles in the coastal environment for engineering and habitat purposes in order to implement suitable protection measures (Airoldi et al., 2005; Martin et al., 2005; Moschella et al., 2005; Burcharth, 2007; Firth et al., 2013a). Ecological engineering is a growing area, and the design of "blue-green infrastructure" (Ido and Shimrit, 2015) is a trend generating attention from academic researchers and industries.

This introductory chapter provides a review on relevant literature to this thesis and outlines the need for the research undertaken. The thesis is broken into two parts: 1) 'Engineering for Ecology' which consists of empirical Chapters 2 to 4; and 2) 'Ecology for Engineering' explained by Chapter 5. The first half of this introductory chapter starts by detailing the needs for coastal protection, particularly with current environmental changes and growing economic pressures. It then outlines the various methods of coastal management, including reviews of the different types of CDSs and their associated functions, and current related policy drivers. Following this is a review of the role of artificial structures as surrogate habitats, including urban ecology, driving factors influencing coastal communities, and relevant research focusing on artificial habitat enhancement methods. This then leads to a review of the role of coastal ecology in the natural environment, coastal protection and potential management schemes. Knowledge gaps in existing research are identified and highlighted, and the chapter concludes with the aims, objectives and rationale for the research undertaken in this thesis (see Table 1.1 for overview).

1.2 The Need for Coastal Protection

Coastal areas are dynamic and complex systems lying on the boundaries between land and marine interactions. They are therefore affected by changes to both environments independently and the interactive impacts between the two such as erosion, storm surges and flooding (Jolley, 2008; Nicholls et al., 2015). Changes in sea-levels and shoreline erosion have been a consistent pressure on coastal areas. There has historically been issues with flooding, a need to protect high value land areas and provide navigation structures for maritime transport (Charlier et al., 2005); however, the need for coastal defence structures (CDSs) and their subsequent designs has developed over time as a response to environmental and socio-economic needs (Nicholls et al., 2008).

Climate change is a major driver in the twenty first century changing environment (Nicholls and Mimura, 1998; Root et al., 2003; Tromp and Van Wesenbeeck, 2011; Jackson and McIlvenny, 2011; Philippart et al., 2011), and coastal areas are the most sensitive to these impacts. One of the largest issues resulting from climate change is the rise in sea levels and storm surges, particularly in tropical areas. Sea level rise is a serious concern for coastal regions including many low lying countries and major cities that will be susceptible to flooding and an ensuing loss of land area (Nicholls and Mimura, 1998; Airoldi et al., 2005; Burcharth et al., 2007; Philippart et al., 2011; Nicholls et al., 2015). It is predicted that sea levels could rise by 0.3 m up to a potential 1.9 m by 2100 (Turner et al., 2009; Vermeer and Rahmstorf, 2009; Jevrejeva et al., 2010; Jackson and McIlvenny, 2011) and have previously increased globally on average by 0.19 m from

Table 1.1: Background themes reviewed in Chapter 1

	Theme	Relevant thesis question	Pages
Section 1.2:	**The Need for Coastal Protection**		**2**
	Overview of the problem and demand for CDS.	How common are CDS now and in the future?	
Section 1.3:	**Environmental Impacts of Coastal Defence Structures**		**5**
	Environmental impacts and pressures associated with CDS.	How can we reduce anthropogenic disturbance and pressures on the natural environment?	
Section 1.4:	**Coastal Management and Policy**		**7**
	Current coastal management strategies and environmental protection.	Is there a future for ecological enhancement policy? Why is this research important in a policy framework?	
Section 1.5:	**Coastal Defence Structures as Habitats**		**12**
	Previous research on CDS as artificial habitats and environmental enhancements.	Curent understanding of differences between natural and artificial shore communities and driving factors. Where does this research fit in?	
Section 1.6:	**Ecology for Engineering**		**17**
	Previous research on ecosystem engineers, particularly in the intertidal environment.	Where does this research fit in and how can we use ecosystem engineering species in coastal protection?	
Section 1.7:	**Knowledge Gaps for New Research**		**19**
	Gaps in current research that need answering.	What are the research gaps and where does this work fit in?	
Section 1.8:	**Thesis Outline**		**20**
	Chapter overviews and what the expected outcomes will be.		

1901 to 2010 (IPCC, 2014; Le Cozannet et al., 2014). Increased sea level causes higher velocities of tidal waves due to the effects of water depth and shoaling (Pethick, 1993; Denny and Gaylord, 2010; Jackson and McIlvenny, 2011). In addition, the possibility of more frequent storm surges as a result of changing weather patterns, combined with the associated coastal compositions and declining sediment supply, is increasing erosion of the coastline and causing shores to become steeper and retreat (Le Cozannet et al., 2014). This has been noted in a study by Taylor et al. (2004) showing that in England and Wales >60% of intertidal zones have become significantly steeper over recent decades. Steepened coastlines will have important effects on wave interactions and propagations along the shores, as a steeper shore results in a higher reflection of wave energy (Jackson and McIlvenny, 2011). This will subsequently lead to severe deterioration of the coastlines and may cause an increase in the transport of sediment offshore (Hoefel and Elgar, 2003; Jackson and McIlvenny, 2011). Such modification and potential loss of land and habitats, adds supplementary pressure on habitat availability and coastal resources.

The need for coastal protection does not only stem from climate change related drivers. Whilst climate change is a major issue adding significant additional pressures to coastal areas, there are other socio-economic and environmental drivers as outlined by Nicholls et al. (2008). With a large proportion of the global population living close to coastal zones, there is an increase in socio-economic activities within these areas (Ido and Shimrit, 2015). Coastal areas are desirable places to live, work and visit (Glasby and Connell, 1999), resulting in increased economic development and urbanisation of the coastline. This subsequently creates growing socio-economic vulnerability (Celliers, 2015). Erosion and flooding in developed areas has resulted in loss and severe damage to economic assets such as infrastructure (e.g. railways and roads), properties, local businesses and arable land (Jolley, 2008). There are also threats to the natural coastal ecosystems and habitats (Dugan et al., 2011) due to significant topographic changes to hard shorelines through erosion and inundation. Hard-rock accounts for approximately 42% of British coastlines, and 80% of global oceanic coastlines, most of which are backed by cliffs (Jackson and McIlvenny, 2011). According to the European Commission CORINE (Co-ordination of Information on the Environment) programme, only 55% of European coastlines (total length of 56,000 km) are thought to be stable, while 19% suffer from erosion problems, and 8% to be depositional (EEA, 1994; Airoldi et al., 2005). Therefore, there is the potential for drastic consequences to our shoreline if no mitigating actions are undertaken.

Hard defence structures have and still are a common approach to defend the coast and actively claim land areas (Charlier et al., 2005; Nicholls et al., 2013). There is now additional pressures to protect against accelerating coastal erosion due to increasing sea level rise and extreme weather conditions. Many coastlines globally are becoming dominated by artificial shores. England alone has 46% of its entire coastline modified (18% UK) (see Table 1.2) (Masselink and Russell, 2013). Europe as a whole has 14% of its coastline now artificial, although that is not evenly distributed across the EU (MAFF, 2000; Moschella et al., 2005; Liquete et al., 2013; Masselink and Russell, 2013), and in some parts of countries such as the Gold Coast and New South Wales, Australia; North Carolina and Louisiana in the USA; and Japan, large extents of coastlines have been replaced or reinforced (Masselink and Russell, 2013; Dafforn et al., 2015). It is therefore essential to increase understanding of the impacts of CDSs on the natural environment in order to develop successful coastal management and adaptation strategies which include stronger and more sustainable CDSs (Moschella et al., 2005).

Table 1.2: Percentage of coastlines in the UK that have been modified to hardened artificial structures. (Data adapted from Masselink and Russell, 2013).

Country	Coastline Modified (%)
UK	18%
Wales	28%
England	46%
Scotland	7%
Northern Ireland	20%

1.3 Environmental Impacts of Coastal Defence Structures

While hard CDS have an important role in protecting our coastlines and assets, they have severe impacts on the natural shoreline and have been the subject of a number of recent reviews (Airoldi and Beck, 2007; Bulleri and Chapman, 2010; Dugan et al., 2011; Dafforn et al., 2015)

1.3.1 Modifications to the Natural Environment

Owing to the role of their engineering function, CDS are often introduced into natural soft-bottom environments resulting in a replacement of the sedimentary habitat by the hard structural footprint. This can alter the dynamics of ecosystems in the area by replacing natural soft habitats with new and foreign habitats (Martin et al., 2005), which open up space for opportunistic species more suited to hard substrates that would otherwise not exist (Chapman and Blockley, 2009). Aside from the inevitable loss of soft habitats, there is also fragmentation of these natural habitats. This can reduce functioning of soft sediment ecosystems. Connectivity of isolated hard-bottomed communities will be increased by creating stepping stones for colonisation and facilitating species expansion (Kimura and Weiss, 1964; Glasby and Connell, 1999; Hawkins et al., 2008; Hoegh-Guldberg et al., 2008; Firth et al., 2013a).

Alongside this, it is possible for hard engineered infrastructure to modify the environmental conditions in the area, such as sediment dynamics, water flow and geomorphology, which can create additional disturbance to natural habitats (Airoldi et al., 2005; Martin et al., 2005; Airoldi and Beck, 2007; Dugan et al., 2008). Vertical structures such as sea walls often reflect wave energy back into the open water, and can therefore create subsequent consequences for adjoining shorelines and communities (Douglass and Pickel, 1999; Bilkovic and Roggero, 2008; Scyphers et al., 2011). Hard structures can also act as physical barriers to natural ecological processes such as coastal retreat resulting in *coastal squeeze*. As a result, there will be a loss of intertidal habitats (Rupp-Armstrong and Nicholls, 2007; Hadley, 2009) and with only sub-optimal, highly stressed habitats remaining in the higher intertidal zone (Haslett, 2000; Kendall et al., 2004; Jackson and McIlvenny, 2011), leading to reduced micro-habitat availability decreasing species diversity and abundance of individual species (Kendall et al., 2004; Jackson and McIlvenny, 2011).

1.3.2 Anthropogenic Disturbance

CDS are designed and calculated to withstand environmental pressures, such as wave action, storm surges and extreme weather (Burt et al., 2011). Nonetheless, this constant activity creates intense pressure on the structures and shoreline, and can result in erosion, scouring, over-topping and undermining (Kamphuis, 2010; Airoldi and Bulleri, 2011; Firth et al., 2012). Over time, this can affect the stability and function of the structure, requiring maintenance to avoid structural failures. Maintenance activity, however, is an exceptionally severe form of ecological disturbance and according to Airoldi and Bulleri (2011) is a critical factor affecting the distribution, abundance and composition of colonising species on artificial structures. It disrupts coastal communities by the abstraction and replacement of areas within the infrastructure where biological settlement has occurred. This subsequently opens up further prospects for opportunistic and invasive species (Dayton, 1971; Sousa, 1979; Hutchinson and Williams, 2003; Tsinker, 2004; Bulleri and Airoldi, 2005; Ruiz et al., 2009; Airoldi and Bulleri, 2011; Mineur et al., 2012), particularly structures supporting transport infrastructure, such as ports and harbours (Lambert and Lambert, 2003; Glasby et al., 2007; Griffith et al., 2009; Dafforn et al., 2009; Rius et al., 2014; Airoldi et al., 2015; Evans, 2016). Anthropogenic disturbance through noise, vibrations and water pollution due to maritime activity; and socio-economic activity increasing waste outputs, emissions and trampling (Burak et al., 2004; Davenport and Davenport, 2006; Phillips and Jones, 2006) are common. In my thesis I focus on maintenance activity as a form of anthropogenic disturbance. I look at previous work been done and the potential areas for reducing the impacts from maintenance activity on the coastal environment.

To date, there has been little research into the impacts of anthropogenic disturbance on the environment and natural ecological communities (Hutchinson and Williams, 2003; Airoldi and Bulleri, 2011) as a result of engineering maintenance activities on CDS. A study by Airoldi and Bulleri (2011) in the north-east Adriatic Sea, Italy, simulated the destruction to colonising communities on CDS by removing species from the face of the structures. The study highlighted significant differences between the recolonisation of intertidal communities in areas where simulated maintenance had taken place, and the times of the year. It was concluded that differences were due to the settlement of invasive species and occupying space that would otherwise be used by dominant native species during spring and summer months, compared to winter.

Not only are the negative impacts of CDS extremely disruptive to the natural environment, the addition of hard infrastructure is costly to implement and maintain. Overall, the impacts of CDS are not a *one-time-only* event, and therefore should be carefully considered by engineers and ecologists when it comes to the design and construction of future coastal protection schemes, as well as by UK, European and international bodies implementing protection laws and legislation.

1.4 Coastal Management and Policy

In the UK there is little formal legislation that requires engineers to pay more attention to the design of protective infrastructure to promote environmental benefits. Bulleri and Chapman (2010) suggest this is one of the main reasons why there has previously been little focus on conservation strategies when designing coastal protection schemes, in contrast to the designs of structures such as artificial reefs (Jensen et al., 2000; Jensen, 2002). However, there is growing acknowledgement by policy makers of the need to modify and enhance structural designs for environmental benefits (HM Government, 2011; Evans et al., 2016). Implementing environmental policies and practice is becoming more evidence based and gaining suitable evidence to push environmental enhancements of artificial structures is difficult due to its interdisciplinary and collaborative nature (Holmes and Harris, 2010; Naylor et al., 2012). Furthermore, practitioners (including engineers, coastal managers, environmental consultants) have different priorities for the implementation of CDS, therefore creating more challenges for deciding suitable legislation (Naylor et al., 2012). Improvements to current coastal management and policies globally are required to ensure that artificial infrastructure is designed for multifunctional purposes. Not only should CDSs protect our growing urban assets and land space, they should also consider and account for our natural assets and minimise damage to the natural environment through optimising designs to reduce structural footprints, resources and maintenance requirements; and promoting ecosystems, biodiversity and social benefits (HM Government, 2011).

"*Integrating biodiversity values into all planning processes*" is a specific objective which has been outlined following the Convention on Biological Diversity (CBD) Conference of Parties (2010) (Naylor et al., 2012). There are various UK, European and international regulations that are aimed to protect and mitigate damage to natural environments, particularly areas which are deemed high priority and under special consideration (see Table 1.3). The policy and legislative material outlined in this section relates specifically to this research (see Naylor et al., 2012, for relevant review of European and UK policies).

Table 1.3: Relevant legislation and policy for the designation and conservation of coastal environments (adapted from Coombes et al., 2011; Naylor et al., 2012).

Level	Legal Framework	Associated Designation
Global	World Heritage Convention (1972)	World Heritage Sites
		Biosphere Reserves
	Ramsar Convention (1971)	Ramsar Sites
European	Convention on Biological Diversity (1992)	Priority Habitat (UKBAP)
		Priority Species (UKBAP)
	EC Biodiversity Strategy (2011)	
	EC Water Framework Directive (2000/60/EC)	
	EC Directive on EIA (85/337/EEC) and (97/11/EEC)	
	SEA Directive (2001/42/EC), and 2004 UK Regulations	
	The Marine Strategy Framework Directive (2008/56/EC)	
	Habitats Directive (1992/43/EEC)	Special Areas of Conservation (SACs)
	Birds Directive (1979/409/EEC)	Special Protection Areas (SPAs)
UK	UK Marine Policy Statement (MPS, 2011)	
	Planning Policy Statement 9 (Biodiversity and Geological Conservation, 2005)	

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Level	Legal Framework	Associated Designation
	Planning Policy Statement 12 (Local Spatial Planning, 2004)	
	Marine and Coastal Access Act (2009) (England and Wales)	Marine Conservation Zones (MCZ)
	Countryside and Rights of Way Act (2000)	Sites of Special Scientific Intrerest (SSSIs)
		National Nature Reserves (NNRs)
		Area of Outstanding Natural Beauty (AONB)
		Heritage Coast
	Natural England and Rural Communities Act (2006)	
	Biodiversity: The UK Action Plan (UKBAP, 1994)	
	Environment Act (1995)	
	Harbour Revision Orders (Harbours Act, 1964)	

Internationally, specific marine conservation commitments have been outlined under the Convention on Biological Diversity and the OSPAR Convention, which aim to protect and preserve key habitat areas and species. These are particularly important when considering the radical changes to coastlines globally, and efforts required to sustain these environments. In Europe, the Water Framework Directive (WFD, 2000/60/EC) is probably the most influential policy relating to the protection of coastal areas. This legislation specifically sets out relevant targets for water bodies, including heavily modified water bodies (HMWB) such as ports, harbours, and coastlines with hard coastal protection (Naylor et al., 2012). The Marine Strategy Framework Directive (2008/56/EC) is also a good lever as it is a marine-specific directive aiming for Good Environmental Status in European seas by 2020. As a more recent legislation, it has specific targets and monitoring programmes for member states to comply with in order to mitigate the degradation of habitats and adopt corrective measures (Borja, 2006; Naylor et al., 2012). The Habitats Directive (1992/43/EC) and Birds Directive (1979/409/EC) has the potential to help maintain the status of coastal areas. However, this directive does not distinctly cover all intertidal areas, such as mudflats and sandflats not covered by water at low tide. In relation to my thesis, Annex 1 of The Habitats Directive includes the protection of 'reefs', however, this does not cover all reefs but those "*extending from the sublittoral uninterrupted into the intertidal (littoral) zone or only occurring in the sublittoral*" (European Commission, 2007). Some of the key aims from The Habitats Directive also covered in Annex 1 Reef Habitats, are to protect and enhance species populations and ecological connectivity, which opens this legislation up as a potential lever for supporting the use of ecologically enhancements in artificial areas.

Other European legislative tools such as the Strategic Environmental Assessment (SEA) Directive (2001/42/EC) and Environmental Impact Assessment (EIA) Directive (85/337/EEC and 97/11/EEC, respectively) are used more specifically to assess and reduce detrimental impacts of construction schemes before they begin. These are highly effective prevention strategies, however require knowledge and understanding of the specific structure-biota interactions, which in coastal environments is hard to predict and can differ largely depending on local area conditions. In the UK, EIAs are used to help policy-makers make rationale decisions regarding the permissions of engineering projects. These processes require supporting evidence, impact reviews and stakeholder engagement to decide whether the proposed project plans are deemed environmentally suitable (Eccleston and Doub, 2012), and have socio-economic and cost benefits.

In the UK, The UK's Marine Policy Statement (UK Parliament, 2011) recognises the need for proactive planning of hard artificial infrastructure. This is also reflected in the Marine and Coastal Access Act in England and Wales, where one of its primary aims is to conserve and enhance (the beauty of) the marine natural environment (Naylor et al., 2012). In order to plan for each area and the specific conditions, the UK coastal management strategies are broken down into 'management units' and outlined in the Shoreline Management Plans (SMPs). These strategies, where appropriate, include the need for added ecological features on coastal protective structures. The types of management strategies that are used within the SMP are outlined in Table 1.4. The most environmentally friendly coastal management method would be to carry out no active intervention in any area and allow the system to develop its own natural shore, or replace it with natural material (Chapman and Underwood, 2011). However, as a result of no action, there would be huge loss of natural and economic assets due to erosion and flooding. Therefore, in most cases the SMP recommends a *hold the line* approach (Environment, 2010) using hard defence structures such as sea walls, embankments, breakwaters and groynes, to name a few.

Table 1.4: Management strategies outlined in the SMP (second generation) for maintaining UK coastlines (adapted from DEFRA, 2006).

Management Policy	Action
No Active Intervention	No schemes or investments planned on the coastal area, therefore allowing it to continue as it is, whether that is through natural process, or using previously constructed CDS.
Hold the Line	Using current or new CDS to retain the position of the shoreline. This may require new methods of defence.
Managed Realignment	Controlled movement of the shoreline, allowing it to retreat naturally but manipulated movement to ensure it retreats in a certain area.
Advance the Line	Construction of new defences on the seaward side.

In the twenty first century, whilst there are many of the same trends driving the development of CDS as during the early days of their construction, there is now growing socio-economic pressures and better environmental awareness Nicholls et al. (2008). The changes in drivers requires close inspection of coastline, their assets and coastal management plans to ensure adequate protection in areas that require it. Climate-induced and non-climate-induced drivers globally will require reviews of coastal zones and will likely see an increase in CDS in some areas susceptible to continued sea level rise, flooding and erosion events. It is also likely that some coastlines may see a decline and abandonment of hard artificial structures where they are no longer serving a functioning role or sustainable to maintain, for example areas in the UK where CDS are situated in front of cliffs will likely be abandoned due to changing shoreline management plans.

Traditionally coastal protection has been implemented through hard CDS because they are seen as permanent and immediate solutions. Some of the main benefits of using hard engineering techniques are immediate protection; resistance against large environmental forces; reduced coastline erosion; and the ability to function in different environments. However, hard engineering techniques are also expensive, require regular maintenance work, create high environmental impacts and even increase erosion of the shoreward bed in some cases (South East Coastal Group, 2012). Consequently, soft engineering approaches, such as beach replenishment, sand dune stabilisation, managed retreat/ realignment and ecological engineering (see Section 1.6) are regarded as more environmentally friendly methods than using hard engineering structures for flood and coastal erosion risk management (FCERM) (Turner et al., 1998; Govarets and Lauwaert, 2009; Temmerman et al., 2013; Hanley et al., 2014; Evans, 2016). Soft engineering refers to working with natural coastal processes and utilising ecological principles and practices to stabilise shorelines. These approaches are becoming more commonly used, particularly in Europe where there is more policy concerned with coastal landscape, habitats and species conservation (Coombes et al., 2009; Naylor et al., 2012). In addition, changing coastal management plans and legislation may see an increase and adoption of new innovative multifunctional approaches (e.g. Swansea Tidal Lagoon) which include soft engineering concepts.

1.5 Coastal Defence Structures as Habitats

Infrastructure placed in any natural environment will inevitably become colonised by primary settlers such as epibenthic marine organisms and biofoulers (Evans et al., 2016), but if left long enough structures could have the potential to develop as functioning ecosystems. Artificial structures are well-known to support organisms homologous to nearby rocky shores (Southward and Orton, 1954; Hawkins et al., 1983; Chapman, 2003; Chapman and Bulleri, 2003; Pinn et al., 2005; Moschella et al., 2005; Hawkins et al., 2016). However, most research suggests that CDS are poor replacements for natural rocky shores (Caceres et al., 2002; Moschella et al., 2005; Carvalho et al., 2013; Deepananda and Udayantha, 2013) and support a lower diversity of species (Chapman, 2003; Pinn et al., 2005; Moschella et al., 2005; Pister, 2009; Firth et al., 2013c; Hawkins et al., 2016; Aguilera et al., 2014; Evans et al., 2016) and different relative abundances of taxa (Connell and Glasby, 1999; Chapman and Bulleri, 2003; Knott et al., 2004; Bulleri et al., 2005; Moschella et al., 2005; Pinn et al., 2005; Pister, 2009; Burt et al., 2010, 2011). Artificial structures often support higher numbers of opportunistic species that colonise them (Bulleri et al., 2005; Airoldi and Bulleri, 2011; Salomidi et al., 2013). They can also provide stepping stones for native (Hawkins et al., 2009; Philippart et al., 2011; Firth et al., 2012; Hawkins et al., 2016) and non-native (Bulleri and Airoldi, 2005; Airoldi et al., 2015) species advancing with climate change. The differences between natural and artificial communities are likely for a number of reasons, such as:

1. **High energy environments** Exposure to high energy environments reduces the number of species that are well adapted and hardy enough to survive (Moschella et al., 2005; Burcharth et al., 2007; Vaselli et al., 2008; Pister, 2009; Firth et al., 2013b). In addition, harsh energy environments provide constant pressure on hard structures and increase erosion, scouring by soft sediment material (e.g. sand, gravel, shingle, cobbles), overtopping and undermining (Kamphuis, 2010; Airoldi and Bulleri, 2011; Firth et al., 2013b).

2. **Shoreline gradients** Artificial shores often (but not always) have drastically steeper gradients and are smaller in size, therefore limiting potential habitat space particularly on the lower-shore (Moschella et al., 2005; Chapman and Underwood, 2011; Perkol-Finkel et al., 2012; Hawkins et al., 2016). Chapman (2003) noted a

significantly reduced proportion of mobile species found on the vertical seawalls in Sydney harbour, compared to adjacent natural rocky shores with a lower shore gradient.

3. **Position on the shore** The role of CDS in protection requires them to be situated in the optimum locations on the shoreline to reduce wave impacts. This often makes their placement higher up the coast than natural shores, therefore limiting their inundation and increasing species exposure levels to desiccation and predation (Moschella et al., 2005; Hawkins et al., 2016). Species vulnerable to desiccation will be less likely to survive in such conditions, and those that are will be faced with higher competition for optimal habitats (Haslett, 2000; Kendall et al., 2004; Jackson and McIlvenny, 2011).

4. **Age**: Natural shores in general are much older in comparison to CDS. Their presence in the marine environment has been long established, and undergone hundreds or years (and more) of weather events and erosion, creating complex formations along the shores which provide important niche habitats and refuge spaces for species (Coombes et al., 2011). Alongside this, the gradual processes on natural shores has provided species with sufficient time to adapt to changing environments and establish past the primary settlement stages to develop diverse assemblages (Carvalho et al., 2013). Some research has been carried out assessing whether age of shores is a significant factor distinguishing natural and artificial structure communities (Bacchiocchi and Airoldi, 2003; Airoldi et al., 2005; Burt et al., 2011). Burt et al. (2011) compared six engineered breakwater structures of varying ages (1-31 years old) with natural reefs to assess differences in the coastal community developments. The results showed distinct variations between artificial and natural habitats in terms of community compositions, but indicated that benthic communities develop to become more similar to natural assemblages as they aged.

5. **Porosity**: Natural shores comprise solid structures. CDS, on the other hand, are constructed in a modular formation using several units to produce a structure, therefore adding porosity to the structure. This modularity and mobility adds weakness to the structure through displacements and breaking during strong storm events, and potential failure of the structure over prolonged periods (Airoldi and Bulleri, 2011). Porosity also provides additional spaces and enables the passage of water through the structure which would not otherwise occur unless it was subjected to thousands of years of erosion.

6. **Habitat complexity**: Several studies have been carried out looking at habitat complexity between natural and artificial shores. Research has shown that CDS typically show low levels of habitat complexity, compared with natural shores (Chapman, 2003; Moschella et al., 2005; Aguilera et al., 2014; Evans et al., 2016; Hawkins et al., 2016). Carvalho et al. (2013) suggests this may be due to higher topographic complexity on natural shores because they have a wider range of three-dimensional structures such as rock pools, overhangs and crevices. This will be explained in further detail below.

7. **Habitat isolation**: Other research has suggested that the deficiency in community diversity on artificial structures may also be due to potential isolation from other nearby shores or created through division from sedimentary habitats creating 'oases' (Davis et al., 2002; Dethier et al., 2003; Airoldi et al., 2005; Perkol-Finkel et al., 2012; Ferrario, 2013; Evans et al., 2016). Although this may provide additional pressures to coastal communities, in my thesis, it is not viewed as a primary factor determining differences between natural and artificial communities because isolation can occur for any both natural and artificial shore types.

In terms of my thesis, focus will be directed to structural complexity as this is viewed as one of the leading differentiating factors between natural and artificial shores, and one that may be manipulated in future to help improve biodiversity on artificial shores without compromising structural integrity. Structural complexity relating to this research is defined as "the physical architecture of a structure" (McCoy and Bell, 1991) which considers the surface roughness and relative abundance of different features such as crevices, holes, overhangs. The concept of adding ecological principles to urban coastal infrastructure is relatively new (Mitsch, 1996; Bergen et al., 2001; Mitsch and Jørgensen, 2003; Odum and Odum, 2003; Perkol-Finkel et al., 2012). Research to date has developed novel concepts for incorporating ecological value to CDS, however, much of this is still viewed as research concepts and lacking practical applications and incorporation into CDS schemes.

1.5.1 Structural Complexity

Structural complexity is frequently identified as a key driver which can influence community composition on natural and artificial structures (Chapman, 2003; Moschella et al., 2005; Carvalho et al., 2013; Aguilera et al., 2014; Evans et al., 2016; Hawkins et al., 2016). A number of studies have been carried out looking at the variations in structural complexity from a micro (<1 cm) up to the macro scale (1- 100 m) (see Appendix A for an overview of studies), and how they impact species colonisation on artificial shores.

Factors such as surface roughness and porosity of construction materials (on a micro scale) (Allen, 1998; Moschella et al., 2005; CIRIA, 2010; Borsje et al., 2011; Perkol-Finkel and Sella, 2015; Coombes et al., 2015); size and placement of the structure on the shore (Moschella et al., 2005; Hawkins et al., 2016); unit size and orientation (Moura et al., 2008; Firth et al., 2012; Carvalho et al., 2013) have all been shown to play an important role in species compositions. Carvalho et al. (2013) notes that natural shores have higher topographic complexity featuring niche micro-habitats such as rock pools, overhangs and crevices, which are essential for species protection and refuge during low tide or strong wave conditions (Chapman and Blockley, 2009), and thus leads to higher biodiversity. However, many CDS offer alternative refuge features through their structural porosity. It is common for CDS to be built using rock armour/ rip rap units which offer engineering functions by reducing wave transmission, reflecting incident waves from the shores, and dissipating wave energy (Dalrymple et al., 1991; Losada et al., 1995; Garcia et al., 2004; Burcharth et al., 2015); and provide smaller environmental footprints (Koraim and Rageh, 2013). In addition, the porous nature of these structures (on a macro scale) provides internal cavities and potential refuge spaces which have not previously been considered or studied as potential habitat areas. Furthermore, the use of porous structures may provide additional protection from unavoidable anthropogenic disturbance such as maintenance works. Internal cavity areas may be less susceptible to wave exposure and strong surges, thus limiting the extent of required maintenance works to external rip rap units. With this in mind, it is important to consider the multifunctional role CDS can play by providing engineering functions and supporting ecological communities through ecologically enhancing design modifications.

1.5.1.1 Built in Modifications

There have been a number of studies which have focused on ecological enhancements, novel designs and new technologies for increasing structural complexity and creating *multifunctional structures* through built in modifications (see Appendix A for an overview of studies). Attempted replication of weathered and water-retaining features such as holes, grooves and rock-pools have shown successful increases in biodiversity and abundance of species on artificial surfaces with these features, compared to artificial surfaces without (Chapman and Blockley, 2009; Martins et al., 2010; Browne and Chapman, 2011; Firth et al., 2012; Perkol-Finkel and Sella, 2015; Ido and Shimrit, 2015; Evans et al., 2016). This is well demonstrated by the BIOBLOCK, a concrete cast experimental unit, designed and deployed in 2012 by the 'Innovative technologies for safer European coasts in a changing climate' (THESEUS) project (Firth et al., 2012). The BIOBLOCK included a variety of modifications on each face to demonstrate the effectiveness of habitat-enhancing features (see Appendix A). The idea behind the BIOBLOCK was a pre-cast unit which could be incorporated into CDS designs, such as revetments and sea walls, from the early construction stages (Firth et al., 2012). Firth et al. (2013c) highlights the importance of water retaining features, particularly on artificial structures, where low habitat heterogeneity limits species richness. In addition, Evans et al. (2016) presented critical findings which demonstrate the successful enhancement of CDS via simple drilled bore holes to replicate rock pools on artificial surfaces. Similar studies which have added pits, crevices and other modifications to CDS and have also demonstrated higher diversities of species (Moschella et al., 2005; Moreira et al., 2007; Chapman and Blockley, 2009; Martins et al., 2010; Chapman and Underwood, 2011). The majority of these methods have been implemented through manual modification post construction. Technologies such as the BIOBLOCK, ECOncrete®) and the Vertipools have created design modifications through casting technology with design features built in. This enables designs to be included from an early stage and saves manual labour and added costs, and is a useful technique to include in future schemes. However, this method still comes with limitations to the extent of the features possible and their complexity. Other technologies such as additive manufacturing (3D printing), an increasingly popular technique in a number of industries, could remove these design limitations. 3D printing enables designs to be incorporated from a digital stage and can create complexity throughout a structure due to its layered manufacturing process, therefore enabling structural features such as cavities and crevices to be added throughout the units, not just on external features as is limited with other previously tested methods. However, in order to consider this technology within coastal engineering, suitable materials are required to meet engineering and environmental standards.

1.5.1.2 Material Selection

Selection of suitable materials is a pivotal aspect in the construction of CDS to ensure engineering functionality, but can also impact the colonisation of epibenthic organisms due to its surface texture and acidity. Most CDS are produced from materials which are generally hard to meet engineering standards, however, these materials generally have high acidity levels, intense carbon footprints and low topographic complexity, such as cast concrete and quarried rock (e.g. granite and Portland limestone) (Allen, 1998; CIRIA, 2010; Coombes et al., 2015; Ido and Shimrit, 2015). Materials with rougher surfaces, such as limestone and sandstone (generally softer materials), have demonstrated higher diversities of species and more complex assemblages than harder, less complex materials which are more frequently selected by engineers, such as granite (Coombes et al., 2011). This is due to the increased surface heterogeneity often created through bioerorion and weathering events providing spaces for primary settlers and euendolithic organisms (boring) (Burcharth, 2007; Burt et al., 2009a; Coombes et al., 2011; Firth et al., 2012), such as algal and bacteria species. Coombes et al. (2011) also noted that while limestone and sandstone revealed higher species diversity, granite showed a higher colonisation of opportune micro-organisms but little biological activity on the surface due to its hard composition. This can reduce the number of later successional organisms able to settle and inhabit the material.

Other research has looked at modifications to traditional engineering methods to make the composition of the substrate more environmentally friendly. ECOncrete® have developed a concrete mixtures with lower alkalinity in comparison to Portland cement (a common maritime material) to provide a more ecologically sensitive material to encourage colonisation (Perkol-Finkel and Sella, 2011). The company also designed units, reef pools, pile casings and wall claddings using their novel concrete mixture to enhance the biological and ecological value of maritime infrastructure and reduce the ecological footprint (see Appendix A). Others have created new concrete mixtures using natural aggregates such as oyster shells and fly ash in structures such as the RECIF project, Robert and Loren (2009); Risinger (2012); Furlong (2012) and Scyphers et al. (2015). These techniques, particularly bivalve shells, provide a more environmentally sensitive solution whilst retaining engineering standards, and have demonstrated faster colonisation and higher diversities of species compared to traditional concrete mixes (Perkol-Finkel and Sella, 2011; Risinger, 2012).

To date, most research on ecologically sensitive designs and multifunctional structures has been carried out on submerged artificial reefs which experience very different environmental pressures (Harris, 2003; Jackson and Tomlinson, 2012; Mendonça et al., 2012; Scyphers et al., 2015). Interest is now shifting to research on the multifunctional potential of structures in the intertidal regions (Chapman and Blockley, 2009; Martins et al., 2010; Browne and Chapman, 2011; Firth et al., 2012; Perkol-Finkel and Sella, 2015; Ido and Shimrit, 2015; Evans et al., 2016), which receive higher environmental disturbance due to their location within the terrestrial-marine boundary (Burt et al., 2011). Developed legislation relating to ecologically sensitive designs and increasing evidence of practical applications is vital to generate support from stakeholders and ensure these methods are applied to future infrastructure schemes (Evans et al., 2017).

1.6 Ecology for Engineering

In order to explore 'Ecology for Engineering', this section will focus on the use of ecosystem engineers as a potential coastal protection method. Many engineering approaches adopt principles from natural coastal protection seen globally in order to enhance resilience. Ecosystems such as coral reefs, mangroves, salt marshes, kelp forests and sea grass beds have been long established as successful natural coastal protection due to their ability to dissipate wave energy and provide habitats for coastal species (Kobayashi et al., 1993; Koch et al., 2006; Anderson et al., 2011; Borsje et al., 2011; Rosman et al., 2013; Anderson and Smith, 2014; Hu et al., 2014). This protective functionality is due to *ecosystem engineers*; organisms that "modify, maintain or create habitats" (Jones et al., 1994) through their ability to self-design and/ or self-organise (Mitsch, 1996; Bergen et al., 2001; Odum and Odum, 2003). Utilisation of ecosystem engineers in planned coastal protection schemes has the potential to provide more environmentally friendly and cost effective methods than hard CDS. However, despite much research in this area there is still a lack of practical applications to determine the feasibility of successfully transplanting ecosystem engineers and their ability to withstand high intensity environments. Research to date has focused on the properties of independent species and their ability to manipulate their surrounding environment to improve conditions. Species such as *Mytilus edulis* (common mussel), *Ostrea edulis* (native oyster), *Crassostrea gigas* (Pacific oyster), *Zostera marina* (common eelgrass), *Ammophila arenaria* (marram grass) on dunes, corals and mangroves have been extensively studied and demonstrate numerous benefits (see Appendix B for review of relevant studies).

Wave energy loading onto coastlines creates high levels of damage to infrastructure and disturbance to ecological species. In Britain, species such as *M. edulis*, *O. edulis*, *C. gigas*, *Z. marina* and *A. arenaria*, can interact with wave dynamics while increasing habitat complexity, structural stability and trap sediment (Meyer et al., 1997; Piazza et al., 2005; Fernando et al., 2008; Borsje et al., 2011; Ysebaert et al., 2011a; Carus et al., 2016). Reef building species, such as mussels and oysters, form tightly packed communities offering natural hydraulic properties by creating turbulence to oncoming waves and reducing energy transfer (Borsje et al., 2011). Large beds can significantly reduce hydrodynamic energy by disrupting the boundary-layer, forming eddies around the structures and creating drag, therefore potentially allowing a reduction in the size of defence structures (Commito and Rusignuolo, 2000; Fernando et al., 2008; Borsje et al., 2011). Ysebaert et al. (2011a) found that tidal wetland vegetation such as *Spartina alterniflora* and *Scirpus mariqueter*, also reduce waves heights by up to 80% over short distances (<50m), therefore decreasing hydrodynamic energy. Similarly, coral reefs possess the same ability to reduce wave energy into the shore (Nelson, 1996). This is due to the rugosity of the natural structure they form which disrupts the flow and therefore reduces energy Fernando et al. (2008). As an added benefit, these species can provide sediment accretion (Meyer et al., 1997; Koch et al., 2006, 2009; Borsje et al., 2011); increase habitat heterogeneity in coastal environments, providing niche micro-habitat spaces for other coastal species, such as rock pools, cracks and crevices; and provide protection against weathering and erosion (Naylor and Viles, 2002; Coombes et al., 2011; Coombes and Naylor, 2012).

Climate change and urbanisation are decreasing natural coastal protection and the associated coastal assemblages. UKNEAFO (2014) noted in the UK alone, 60% of the 682 recorded coastal species have declined over recent years, and 29% have severely declined. Costanza et al. (1997) estimated that coastal regions contribute 77% of global ecosystem services. Ecosystem services provide multidimensional values, including monetary values and other more qualitative measures (UKNEAFO, 2014). It is estimated that the economic value of coral reefs globally reaches approximately \$29 billion, of which coastal protection accounts for \$9 billion and biodiversity \$5.5 billion (Cesar et al., 2003; Conservation International, 2008). Similarly, Brander et al. (2006) showed that wetland areas provide approximately \$2800 per hectare per year in economic value. In the UK, where there is a diverse range of natural coastal protection, there are very few, if any, valuation studies carried out to determine the specific economic value provided through protection of UK coastal habitat services (see UKNEAFO, 2014, section 4.S.6.2). However, it is still acknowledged that the role of these areas is of high importance to UK coastal areas.

1.7 Knowledge Gaps for New Research

Extensive review of current literature has highlighted the growing field of ecological engineering, particularly in the coastal environment. Through this review I collated information related to prior research focusing on the development of CDS for environmental benefit in order to identify areas of interest and gaps for future research. In addition, working alongside industry sponsors posing real-life engineering questions and issues concerning CDS designs and mitigating environmental impacts, there were a number of clear knowledge gaps which surfaced, particularly relating to the practical application and implementation of enhancement methods to improve sustainability and biodiversity on CDS. Following the review of current literature and identification of future coastal protection needs, I explored potential research topics and suitable methodology in order to clearly define feasible knowledge gaps to be investigated. My thesis addresses four key knowledge gaps that will provide information for engineers and ecologists to develop a better understanding of coastal ecosystem demands, design more sustainable and multi-functional hard CDS, and improve understanding of the benefits of using select ecosystem engineers for coastal protection.

Enhancement of CDS can support higher biodiversity levels and similar communities to natural shores (Chapman, 2003; Moschella et al., 2005; Chapman and Blockley, 2009; Martins et al., 2010; Browne and Chapman, 2011; Firth et al., 2012; Carvalho et al., 2013; Aguilera et al., 2014; Perkol-Finkel and Sella, 2015; Ido and Shimrit, 2015; Evans et al., 2016; Hawkins et al., 2016). Moschella et al. (2005) suggested that modifications to the surfaces of artificial structures would provide greatest enhancement success at the smallest (micro) habitat scale (1 - 100 cm) where this is greatest influence over primary settlers. This prompted further studies by Firth et al. (2014) and Evans et al. (2016) who carried out studies at the <1 cm to 10 m scale. These research projects enhanced successful settlement and colonisation by marine organisms leading to higher biodiversity. The suggestion that the spatial scale for implementing modifications will reflect the species diversity found on artificial shores has prompted my research into the difference between natural and artificial shores at different spatial scales (see Chapter 2). This also opens up questions regarding the multi-functionality of artificial CDS at a structural level. The use of multifunctional structures has been observed previously (Harris, 2003; Jackson and Tomlinson, 2012; Mendonça et al., 2012; Scyphers et al., 2015), however most of these refer to submerged "artificial reefs" which experience very different environmental pressures. Little research has been carried out on the multi-functional potential of structures in the intertidal regions, which receive higher environmental disturbance due to their location within the terrestrial-marine boundary (Burt et al., 2011).

1.8 Thesis Outline

Following the identification of four key knowledge gaps, my thesis aims to build upon previous work investigating the role of CDS for protection and surrogate habitats. This research will focus on two overarching themes *'engineering for ecology'* and *'ecology for engineering'*. Through the integration of both themes, it intends to produce new knowledge and a better understanding of the fundamental characteristics required to design more environmentally sensitive CDS for promoting biodiversity. Furthermore, it considers the benefits and feasibility of using ecosystem engineers in coastal protection to create more natural and sustainable coastal defence schemes. My thesis will present four empirical chapters and their associated aims and objectives, followed by a concluding chapter discussing the overall findings, relevance and impacts of my research.

1.8.1 Aims and Objectives

The overall aim of this thesis (**Aim 1**) is to assess the sustainability of coastal protection methods in engineering and maintaining biodiversity. Improving the design and function of CDS as suitable habitats for coastal assemblages, requires a better understanding of the key factors influencing settlement and colonisation of coastal species. Furthermore, with the propagation of CDS replacing and fragmenting much of the natural coastlines, there is a demand for more sustainable protection methods. Natural protection is established in many coastlines globally, and promotion of these natural resources could reduce or replace hard defence infrastructure. In order to successfully establish natural engineering methods, understanding of the engineering role of ecosystem engineering species and their tolerance to varying environmental conditions is key. This research aims to provide novel and additional evidence supporting both of these fundamental coastal engineering challenges.

1.8.2 Chapter 2: Comparing Biodiversity and Topography on Natural and Artificial Shores

This chapter investigates the biological and topographic differences between natural and artificial shores along the South Coast of the UK, and explores whether spatial scale effects biological communities and topography. Understanding of the effects of spatial scale on biological communities and topographic complexity can help engineers and researchers to better understand the requirements for successful colonisation and increasing biodiversity, thus design more environmentally sensitive CDS which target key spatial scales. The main findings of this study showed no significant difference in species diversity between natural and artificial shores, spatial scales or due to topography. It did, however, indicate that natural and artificial shores demonstrate different characterising species within the communities. The findings suggest that invertebrate species are better adapted to the harsher environmental conditions generally present where artificial CDS are placed, most likely due to their mobility.

Aim 2: To determine if biological communities and topographic complexity on natural shores differ to those on artificial shores.

- Objective 2a: Determine if biological communities on natural shores differ to those on artificial shores, and does this occur at different spatial scales.
- Objective 2b: Determine if topographic complexity on natural shores differ to those on artificial shores, and does this occur at different spatial scales.
- Objective 2c: Determine whether any difference between biological communities is related to the topographic complexity of the structure.

1.8.3 Chapter 3: Hidden biodiversity in cryptic habitats provided by porous coastal defence structures

Following chapter 3, this chapter investigates the role of porous CDS in habitat provision. Taking advantage of a groyne reduction and the opportunity to access and survey the internal environment of a porous CDS, this chapter considers whether the more benign internal environment provides functional habitat spaces for coastal assemblages. Chapter 3 suggests that invertebrate species are better adapted to survival on CDS, therefore internal and more protected habitat spaces may provide refuge for less hardy species. The results of this study shows that the internal environment does provide functional habitat space supporting higher species richness and diversity than external surfaces.

Aim 3: To asses whether internal environment of porous CDS provides functional habitat spaces for coastal assemblages.

- Objective 3a: Examine species richness and diversity in internal habitats on the porous defence structure compared to external habitats.
- Objective 3b: Evaluate the extent of anthropogenic disturbance caused by coastal engineering work.

Outputs

- Presented and published in the ICE Coastal Management Conference Proceedings 2015 (Sherrard et al., 2015)
- Manuscript Submitted to Journal of Coastal Engineering for Review July 2016 (Sherrard et al.)

1.8.4 Chapter 4: Additive Manufacturing for Coastal Engineering

This chapter investigates the use of 3D printing as an effective engineering tool to design more intricate and complex CDS to increase habitat heterogeneity. Field experiments carried out at Highcliffe, UK, determine the biological capabilities of the novel and more sustainable material created through *D-Shape* 3D printing. The key results of this chapter show successful colonisation of the material, particularly on increased surface roughness. However, the study indicated key issues to be considered in the further development and use of 3D printed material.

Aim 4: To evaluate whether 3D printing is an effective and economic method for adding complexity and ecological value to CDS.

- Objective 4a: Examine the biological colonisation on *D-Shape 'Sorel Cement'* material.
- Objective 4b: Examine the physical characteristics of *D-Shape 'Sorel Cement'* material in a marine intertidal environment.

1.8.5 Chapter 5: Ecological Engineering for Coastal Protection

This chapters explores the second theme in this thesis, *'ecology for engineering'*. Through physical wave studies, this chapter investigates whether ecosystem engineering mimic (e.g. sea grass, sea weed, barnacles, limpets, mussels) can significantly reduce wave velocities. In addition, while previously studied hard structural enhancement features are designed to promote biodiversity on artificial structures, this chapter explores their hydraulic properties. The results showed significant differences between all of the mimic species, with all of them positively reducing wave velocities. There was also a significant reduction in wave velocities due to increased surface complexity.

Aim 5: To determine whether specific characteristics of 1) mimic intertidal and sub-tidal species and 2) structural enhancing features are more successful at reducing hydraulic loading.

- Objective 5a: Compare the wave velocity with and without the presence of different mimic species.
- Objective 5b: Compare the wave velocity with and without the presence of structural enhancement features.
- Objective 5c: Evaluate whether certain species characteristics or surface textures are related to reducing wave velocity.

1.8.6 Chapter 6: General Discussion

In my concluding chapter, I summarise the key findings from my research and their applicability to coastal engineering and management. I then review the challenges experienced and identified through the various research chapters in order to provide recommendations for future studies. I also discuss the academic and industrial impacts of my research, and the success to date it has achieved on the wider engineering and environmental disciplines. Finally, I discuss the knowledge gaps discovered through my research and still remain in order to progress effective implementation of ecological engineering practices within industry and management schemes.

Chapter 2

Comparing Biodiversity and Topography on Natural and Artificial Shores

Abstract

There is a growing demand for more coastal protection of urbanised coastlines made worse by impacts such as erosion and flood risk associated with climate change leading to stormier and rising seas. The replacement of natural shorelines with artificial structures is leading to a severe loss of natural soft-bottom marine habitats, fragmentation of natural areas, and altering biological communities. Artificial shores are generally regarded as poor substitutes for natural rocky shores and support lower species diversity due to a lack of structural complexity limiting habitat availability coupled with high levels of environmental and anthropogenic disturbance. My study compares the biological communities and topography between natural and artificial shores along the South Coast of the UK at different spatial scales (micro $\sim$ 0.10 m, meso $\sim$ 1.0 m, macro $\sim$ 10 m). There was significant difference in biological communities between natural and artificial shores, and independent sites; but not due to topographic complexity. Natural and artificial shores demonstrated different characterising species within the communities. Natural shores were colonised by species such as fucoids and some foliose red algae, while artificial shore communities were largely determined by invertebrate species such as limpets and barnacles. This study provides evidence for the need for including habitat considerations in design of artificial structures, particularly the topographic complexity formed between the rock units that provides potentially more benign habitats to support intertidal species.

2.1 Introduction

The impacts of climate change, such as sea level rise and extreme weather conditions leading to flood risk and erosion (IPCC, 2007a,b; Hoegh-Guldberg and Bruno, 2010; Philippart et al., 2011), are leading to the demand for more hard coastal protection along urban coastlines globally. The growing population in coastal areas (Nicholls et al., 2015) resulting from demand from industry, transportation, business and tourism is leading to increased defences to protect people, assets and natural coastal resources (Airoldi et al., 2005; Airoldi and Beck, 2007; McGranahan et al., 2007; Bulleri and Chapman, 2010). Coastal defence structures are designed to meet local societal needs in a particular environmental context (Burt et al., 2011), therefore they can differ vastly even within nearby areas in terms of size, substrate, porosity, placement along the intertidal/ sub-tidal areas, and orientation on the shore (Moschella et al., 2005; Moura et al., 2008; Firth et al., 2012; Carvalho et al., 2013). Their protective functionality means they are (generally) exposed to higher environmental stresses compared to natural shores, such as extreme wave action, coastal erosion and sediment transport leading to scouring (Moschella et al., 2005; Jolley, 2008; Dugan et al., 2011; Airoldi and Bulleri, 2011). The replacement of natural shorelines with artificial structures is leading to a severe loss of natural soft-bottom marine habitats (Martin et al., 2005), fragmentation of natural areas (Dafforn et al., 2015), and opening up prospects for opportunistic and invasive species to colonise (Airoldi et al., 2005; Hawkins et al., 2009; Philippart et al., 2011; Firth et al., 2012; Hawkins et al., 2016). Alongside this is the additional pressure from anthropogenic disturbance due to the associated maintenance activity and local industries (e.g. ports, harbours) (Airoldi and Bulleri, 2011).

A number of studies have compared the biodiversity (Southward and Orton, 1954; Chapman and Bulleri, 2003; Pinn et al., 2005; Pister, 2009; Firth et al., 2013c; Deepananda and Udayantha, 2013) and structural complexity (Chapman, 2003; Moschella et al., 2005; Aguilera et al., 2014; Evans et al., 2016; Hawkins et al., 2016) on natural rocky shores and artificial shores. Artificial shores are often regarded as poor substitutes for natural rocky shores and support lower species diversity (Caceres et al., 2002; Chapman, 2003; Moschella et al., 2005; Pinn et al., 2005; Pister, 2009; Carvalho et al., 2013; Deepananda and Udayantha, 2013; Firth et al., 2013c; Aguilera et al., 2014; Evans et al., 2016; Hawkins et al., 2016) due to high levels of environmental and anthropogenic disturbance; steeper shoreline gradients; age; position of CDS on the shore; and structural complexity (including topography) (see Section 1.5 for more details). Research has been carried out investigating a number of different enhancement methods that could be implemented at different spatial scales (see Appendix A). Moschella et al. (2005) proposed that modifications to artificial structures would provide the greatest enhancement success at the micro-habitat scale (0.01 1 cm). Further research by Burt et al. (2009b),Coombes et al.

(2011) and Firth et al. (2012) focusing at the micro-spatial scale, show that rough surfaces provide more diverse and complex species assemblages in artificial habitats over time, compared to smooth surface textures that are highly colonised by opportunistic species. Their research only provides indications of complexity at a micro-scale, and did not consider the differences between habitat opportunities available at larger scales through the presence/ absence of micro-habitats. Carvalho et al. (2013) suggests micro-habitats are fundamental to species diversity and that natural shores have a higher topographic complexity because they have a wider range of three-dimensional structures such as rock pools, overhangs and crevices. This was later addressed by Firth et al. (2014) and Evans et al. (2016) who carried out research at <1 cm to 10 m scales. Firth et al. (2014) both compared biodiversity patterns between rock pools and emergent rock and natural shores from Norway and across the UK. It was found that rock pools were more taxon rich than emergent substrata, and algal groups tended to prefer shallower habitats, as opposed to deeper alternatives. Evans et al. (2016) carried out a similar study but on an artificial granite rock breakwater with drilled-core artificial rock pools, and further compared biodiversity patterns between nearby natural rocky shores and the artificial rock pools. Overall, Evans et al. (2016) found different community composition between artificial and natural pools but greater species richness in drilled-core rock pools than adjacent granite rock surfaces on the breakwater, and similar species richness to natural rock pools on nearby rocky shores. In addition, more species were found in shallow drilled-core pools (30% more) than deeper pools. These studies demonstrate the important of complex features at the <1 cm to 10 m scales as opposed to bare, flat and low complexity surfaces.

The impacts of spatial scales on ecological communities and their processes is important for ecologists in order to better understand community drivers and conceptual problems (Underwood et al., 2000). The impacts of spatial scale on ecological communities has long been established to impact dispersion and distribution of species (Wiens, 1989). A number of studies have been carried out looking at the impacts of spatial scales on the abundance and distribution of species within the natural coastal environment, noting differences (Johnson and Scheibling, 1987; Astles, 1993; Metaxas and Scheibling, 1993; Metaxas et al., 1994; Seed, 1996; Hull, 1999; Kelaher et al., 2001; Bussell et al., 2007). Bussell et al. (2007) claims that physical conditions can vary at spatial scales, therefore effecting the complexity of habitats and its associated assemblages. Whilst the variation in ecological communities due to spatial scale is accepted amongst ecologist, it has not yet been considered within the applications and designs of artificial habitats. The impacts of spatial scale on communities colonising artificial structures requires further knowledge in order to better inform engineers when implementing modifications to artificial structures for increased biodiversity. To date, a comparison between biodiversity at different spatial scales on natural and artificial shores has not been carried out, and could provide evidence for improved modifications to artificial structures.

2.1.1 Aims and Objectives

The overall aim of this chapter is to provide more evidence to support the suggestion that spatial scale plays a role in the successful colonisation of diverse biological communities on coastlines. This should then provide further valuable information for engineers and ecologists when designing ecologically sensitive coastal defences. To date, no research has directly compared the biological communities present on natural and artificial shores at different scales. Furthermore, there is little information comparing topographic complexity at different scales and whether it plays an influential role in the colonisation of marine intertidal species. Structural complexity is important for increasing habitat availability and providing niche micro-habitats, which are essential refuge and protection features for species living in the intertidal zones (Chapman and Blockley, 2009). I addressed the following scales: micro $\sim$ 0.10 m, meso $\sim$ 1.0 m, macro $\sim$ 10 m.

Thus in this chapter I tested the following hypotheses:

1. There is a significant difference in biological communities and topography between natural and artificial shores.
2. There is a significant difference in biological communities and topography at different scales.
3. Topographic complexity has a positive correlation with biological diversity but this will vary with scale.

2.2 Methodology

2.2.1 Site Selection

A field survey comparing natural and artificial shores was carried out during July to September 2013. The study sites were located around the Isle of Wight (IoW) and south Dorset coast. Sites were selected based on the following criteria:

1. ‘Natural shores’ were those consisting of rocky intertidal habitats.
2. ‘Artificial shores’ were those constructed using rubble mound coastal defence designs.

3. Proximity to another natural and/or artificial shore site was <5 km.

4. They were situated within the mid-lower intertidal zone and undergo regular full or partial tidal submersion.

5. They were logistically feasible and accessible.

Out of a potential 18 pre-selected sites from the IoW and south Dorset area, seven suitable study sites matched the criteria and were selected, four natural and three artificial shores (Table 2.1).

Table 2.1: Table of natural and artificial sites where field survey were carried out for this study. All artificial shores consisted of rouble mound defence structures over between the ages of 10-50 years old.

Site	Assigned Label	Location	Site Type
Lyme Regis	LR	50°42'54"N, 2°57'7"W	Natural
Fishbourne	FB	50°44'5"N, 1°12'31"W	Natural
Whitecliffe	WC	50°40'28"N, 1°5'16"W	Natural
Seaview	SV	50°43'16"N, 1°6'34"W	Natural
Lyme Regis Harbour	LRH	50°43'8"N, 2°56'8"W	Artificial
Ryde Harbour	RH	50°43'58"N, 1°9'19"W	Artificial
Ventnor Harbour	VH	50°35'34"N, 1°12'21"W	Artificial

2.2.2 Field Surveys

Field surveys sampled the biological organisms at three scales (micro $\sim$ 0.10 m, meso $\sim$ 1.0 m, macro $\sim$ 10 m). To account for the variation in spatial scale, sampling techniques were selected to calculate and compare topographic complexity between levels (see Table 2.2). Topographic complexity was calculated at each spatial scale using a ratio (TC) of known sample length versus the recorded topography to determine a comparable unit between techniques (Frost and Burrows, 2005). Larger ratios imply a greater surface roughness or complexity. Biological samples were taken using quadrats of varying sizes for each spatial scale (see Table 2.2 for details) to calculate the benthic community at each site per spatial scale. For all spatial scale, abundance of all taxa was recorded using the semi-quantitative SACFORN abundance scale (i.e. S = Super Abundant, A = Abundant, C = Common, F = Frequent, O = Occasional, R = Rare, N = Not recorded) (Hiscock, 1996; JNCC, 2015). Taxa were identified and recorded as close to species level as possible. Where species could not be identified *in situ*, photographs or samples were taken for more detailed observations in biological laboratories. For each site, additional physical and environmental information was recorded using set criteria (Table 2.3). All samples at each scale were selected at random along a 50 m span of the mid-low water level on the shore. Positions were determined using a random number generator in Excel. Quadrats were not used if the location randomly selected consisted of >50% sandy substrate or a gap or crevice in the structure which was inaccessible for sampling.

Table 2.2: Topographic and ecological sampling techniques used at each spatial scale

Scale	Topographic Sampling	Biological Sampling
$\sim$ 0.10 m	Profile gauge	0.10 x 0.10 m quadrat sampling
$\sim$ 1 m	Chain and tape	0.50 x 0.50 m quadrat sampling
$\sim$ 10 m	Chain and tape	0.50 x 0.50 m quadrat sampling

2.2.2.1 Micro-Scale: $\sim$ 0.10 m

At the micro-scale ($\sim$ 0.10 m) twenty paired topographic and biological samples were selected at random along a 50 m span of the mid-low water level on the shore. Positions were determined using a random number generator in Excel. A 250 mm long profile gauge with 1 mm thick plastic needles extending 10 mm in length, was selected to measure the topography of 20 x 250 mm profile samples. The profiles were taken from a flat gauge being pressed into the surface of the substrate until all the needles conformed to the shape of the substrate. In the case of ecological material obstructing the profile gauge, this was pushed to the side to expose the substrate. The profiles were traced onto plain A4 paper and the middle 10 cm was selected for the profile sample to avoid any inaccuracies from the ends of the gauge. The length of the recorded topography was calculated with a 0.5 mm piece of thin, highly flexible wire used to trace the profile and then extended and measured using a ruler. The ratio was then calculated by dividing the known length (10 cm) by the recorded topography length. At the same sample location as each of the twenty topographic samples, twenty 10 x 10 cm biological quadrat samples were recorded using the SACFORN abundance scale detailed previously.

Table 2.3: Environmental and physical variables recorded for each of the seven sites from the Isle of Wight and Dorset coast. Variables are adapted from Marine Nature Conservation Review (MNCR) site, littoral habitat descriptors (Hiscock, 1996; Evans, 2016).

Variable	Scale	Notes
Wave Exposure	Very exposed	Prevailing wind and swell onshore
	Exposed	Prevailing wind onshore, offshore shallows/obstructions
	Moderately exposed	Prevailing wind offshore but onshore wind frequent
	Sheltered	Fetch <20 km; offshore shallows/obstructions
	Very sheltered	Fetch <20 km in any direction and <3 km to prevailing wind
Surrounding Habitat	Rocky Mixed Sandy Muddy	Characteristic of direct surrounding habitat (<1 km) by visual inspection: Boulders, Shingle/ Pebbles, Gravel (4-16 mm), Sand (0.063-4 mm), Mud (<0.063 mm), Shells (empty), Artificial
Lowest Shore Height	Littoral fringe Eulittoral Sublittoral fringe	Estimated visually based on strand line and low tide mark
Material	Concrete Granite Limestone Stone (other)	
Micro-habitat Abundance	1 = Low 2 = Moderate 3 = High	Estimated visually based on abundance of Fissures (>10 mm), Crevices (<10 mm) and Rockpools

2.2.2.2 Meso-Scale: $\sim$ 1.0 m

At the meso-scale ($\sim$ 1.0 m) ten paired topographic and biological samples were selected at random along a 50 m span of the mid-low water level on the shore. Positions were determined using a random number generator in Excel. The "chain and tape" method (Luckhurst and Luckhurst, 1978; Frost and Burrows, 2005) was selected to measure the topography of 10 x 1.0 m profile samples using a standard 50 m long industrial measuring tape, and heavy plastic chain with 0.03 m links. The ratio was calculated by dividing the known length (1.0 m) by the recorded topography length using the chain. In the case of ecological material obstructing the profile gauge, this was pushed to the side to expose the substrate. At the same sample location as each of the ten topographic samples, ten 0.5 x 0.5 m biological quadrat samples were recorded using the SACFORN abundance scale detailed previously.

2.2.2.3 Macro-Scale: $\sim$ 10 m

At the macro-scale $\sim$ 10.0 m, six topographic samples were selected from two random locations along a 50 m span of the mid-low water level on the shore. At each of the two locations, three topographic samples were selected each measuring 10.0 m, but laid horizontally, vertically and diagonally along the sampling area. The "chain and tape" method (Luckhurst and Luckhurst, 1978; Frost and Burrows, 2005) measured the topography of the 6 x 10.0 m profile samples as detailed previously. In the case of ecological material obstructing the profile gauge, this was pushed to the side to expose the substrate. Within the two sample areas, ten 0.5 x 0.5 m biological quadrat samples were recorded using the SACFORN abundance scale detailed previously.

2.2.3 Statistical Analysis

Statistical analysis was carried out to determine the differences in biodiversity measured as raw abundance and diversity and topographic complexity between natural and artificial shores at different spatial scale. Biological and topographic data first underwent separate analysis using appropriate tests to determine independent results, and were then analysed together to assess interactions.

2.2.3.1 Community Data

Statistical analyses on the biological data were carried out in PRIMER-E v6 and PERMANOVA statistical software (Clarke, 1993; Anderson et al., 2008; Clarke et al., 2014). Graphs of the mean number of species recorded at each site were first prepared to visualise the raw data, followed by species accumulation curves using PRIMER v6 with 999 random permutations. The SACFRON scale was then converted to numerical scores between 0 and 6 (i.e. S = 6, A = 5, C = 4, F = 3, O = 2, R = 1, Not recorded = 0). Bray-Curtis similarity coefficients (Bray and Curtis, 1957) were calculated using square-root transformed data to stabilise variances (Anderson et al., 2008). Non-metric multi-dimensional scaling (MDS) plots, calculated on a Bray-Curtis similarity matrix of Euclidean distances (Bray and Curtis, 1957), were used to visualise patterns using rank similarities and hierarchical clustering in multivariate data (Clarke, 1993; Clarke et al., 2014) for the factors of shore type (Natural or Artificial), scale (macro, meso or micro) and site (from the seven sites selected).

To test parts of the first and second hypotheses, Permutational Multivariate Analysis of Variance (PERMANOVA), based on 9999 unrestricted random permutations of residuals (Anderson et al., 2008), tested for differences in multivariate species assemblages in response to Shore type, Spatial scale and Site. Factors used in the analysis were: Shore type (fixed, 2 levels: natural, artificial), Spatial scale (fixed, 3 levels: 10m, 1m, 0.1m), and Sites (fixed, 7 levels: Fishbourne, Seaview, Whitecliffe, Lyme Regis, Ryde, Ventnor, Lyme Regis Harbour). To explore differences between biological communities due to the three factors Shore type, Spatial scale and Site, Similarity Percentage (SIMPER) analysis (Clarke, 1993; Anderson et al., 2008; Clarke et al., 2014) was used to identify percentage contributions of individual species providing the dissimilarity between each factor and the interactions within each factor, to investigate their impacts on community compositions. Furthermore, clustering, using the group average linkages and a 1% SIMPROF parameter, was undertaken based on Bray-Curtis resemblance matrix of untransformed data. Visual inspection of the resulting dendrogram aided in further clustering into '*supersets*' to determine ecologically meaningful groups for further analysis using MDS plots and SIMPER to identify the key species within each superset contributing to the similarity of those communities. Prior to multivariate analyses, PERMDISP routine was used to test homogeneity of multivariate dispersions within groups (Anderson et al., 2008; Anderson and Walsh, 2016). Although PERMANOVA is considered more flexible with regard to assumptions than univariate parametric statistics, it makes the implicit assumptions that dispersions are roughly constant across groups. Therefore, PERMDISP can assist in the interpretation of PERMANOVA analysis by testing differences between groups.

2.2.3.2 Topographic Complexity

The topographic ratio results (Surface Rugosity Index) were displayed graphically for each spatial scale and for natural and artificial shores to visualise any obvious patterns. Univariate Analysis of Variance (ANOVA) was used to test for significant differences in the Surface Rugosity Index data at different spatial scales and shore types in order to test the first and second hypotheses. The ratio results were then plotted against the total number of species recorded per site and at each spatial scale. For the micro and meso-scales, a direct statistical comparison between the ecological and topographic data was carried out using PRIMER v6 RELATE correlation analysis (Anderson et al., 2008). Log(X+1) transformation and normalisation of the topographic data were used to create a resemblance matrix of Euclidean distances, before running a RELATE routine using Spearman (ρ) rank correlation analysis to compare the new topographic resemblance matrix and biological resemblance matrix and test the third hypothesis.

2.3 Results

2.3.1 Biological Sampling

Before sampling, environmental and physical conditions were observed and recorded based on the variables outlined in Table 2.3. Natural shores consisted of softer materials than artificial shores. In general they had a surrounding environment consisting of rocky or coarser materials than artificial shores which were mainly surrounded by sandy environments. In total, 24 different species were recorded colonising seven sites across the south coast and Isle of Wight, UK. Initial graphical presentations of the mean number of species counted during the biological sampling across all sites and spatial scale (Figure 2.3) showed no obvious pattern between natural and artificial species present apart from two notable sites on the natural shores, Seaview and Lyme Regis, which had a higher mean count at the 10m and 1m spatial scale levels. Ryde Harbour at the 10m level was the only artificial shore which displayed a higher mean species count. Overall, the natural shore at Lyme Regis had the highest mean number of species at all spatial scales. Figure 2.4 shows species accumulation curves for each site across the three spatial scale: macro ($\sim$ 10 m), meso ($\sim$ 1.0 m) and micro ($\sim$ 0.1 m). The curves across all spatial scale show good plateaus for most of the sites, apart from Lyme Regis natural site that shows steep, growing curves across all three graphs.

Table 2.4: Seven natural and artificial sites, meeting the selection criteria for this study, were surveyed from along the south coast and Isle of Wight, UK.

Site	Assigned Label	Site Type	Wave Exposure	Surrounding Habitat	Lower Shore Height	Material	Micro-habitat Abundance
Fishbourne	FB	Natural	Sheltered	Mixed	Eulittoral	Stone (Other)	Moderate
Seaview	SV	Natural	Moderately exposed	Mixed	Eulittoral	Limestone	High
Whitecliffe	WC	Natural	Moderately exposed	Rocky	Sublittoral fringe	Stone (Barton Sands)	High
Lyme Regis	LR	Natural	Exposed	Rocky	Sublittoral fringe	Stone (Cementstone)	High
Ryde Harbour	RH	Artificial	Moderately exposed	Sandy	Littoral fringe	Concrete	Low
Ventnor Harbour	VH	Artificial	Very exposed	Sandy	Eulittoral	Concrete	Low
Lyme Regis Harbour	LRH	Artificial	Exposed	Mixed	Sublittoral fringe	Granite	Low

Figure 2.1: Four natural sites, meeting the selection criteria for this study. (a) Fishbourne (b) Seaview (c) Whitecliffe (d) Lyme Regis.

Figure 2.2: Three artificial sites, meeting the selection criteria for this study. (a) Ryde Harbour (b) Ventnor Harbour (c) Lyme Regis Harbour

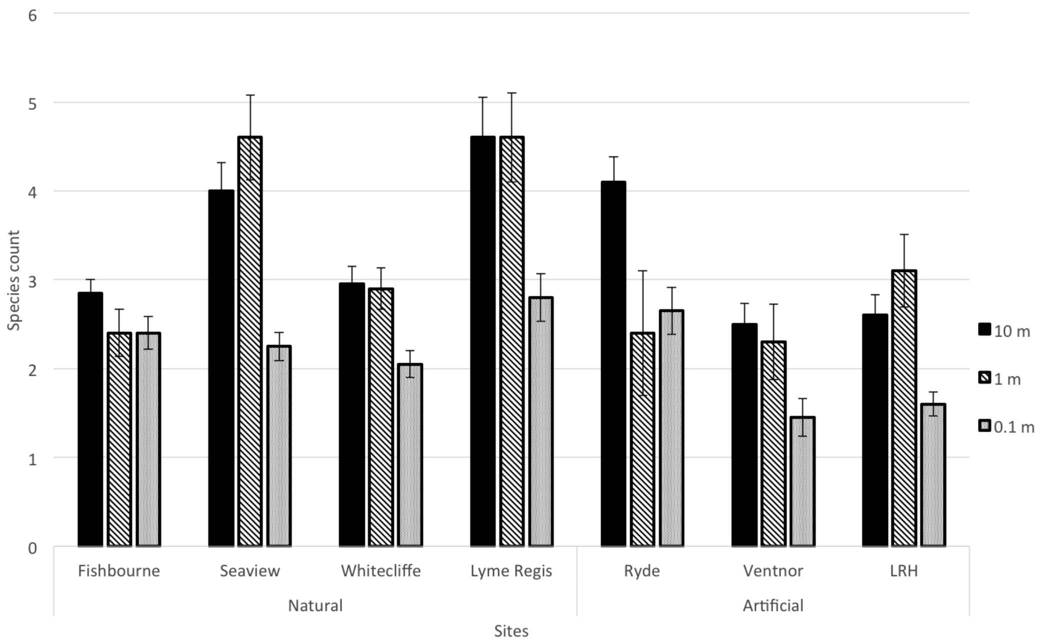

Figure 2.3: Mean species count for each of for each site at different spatial scale with standard error (SE) bars.

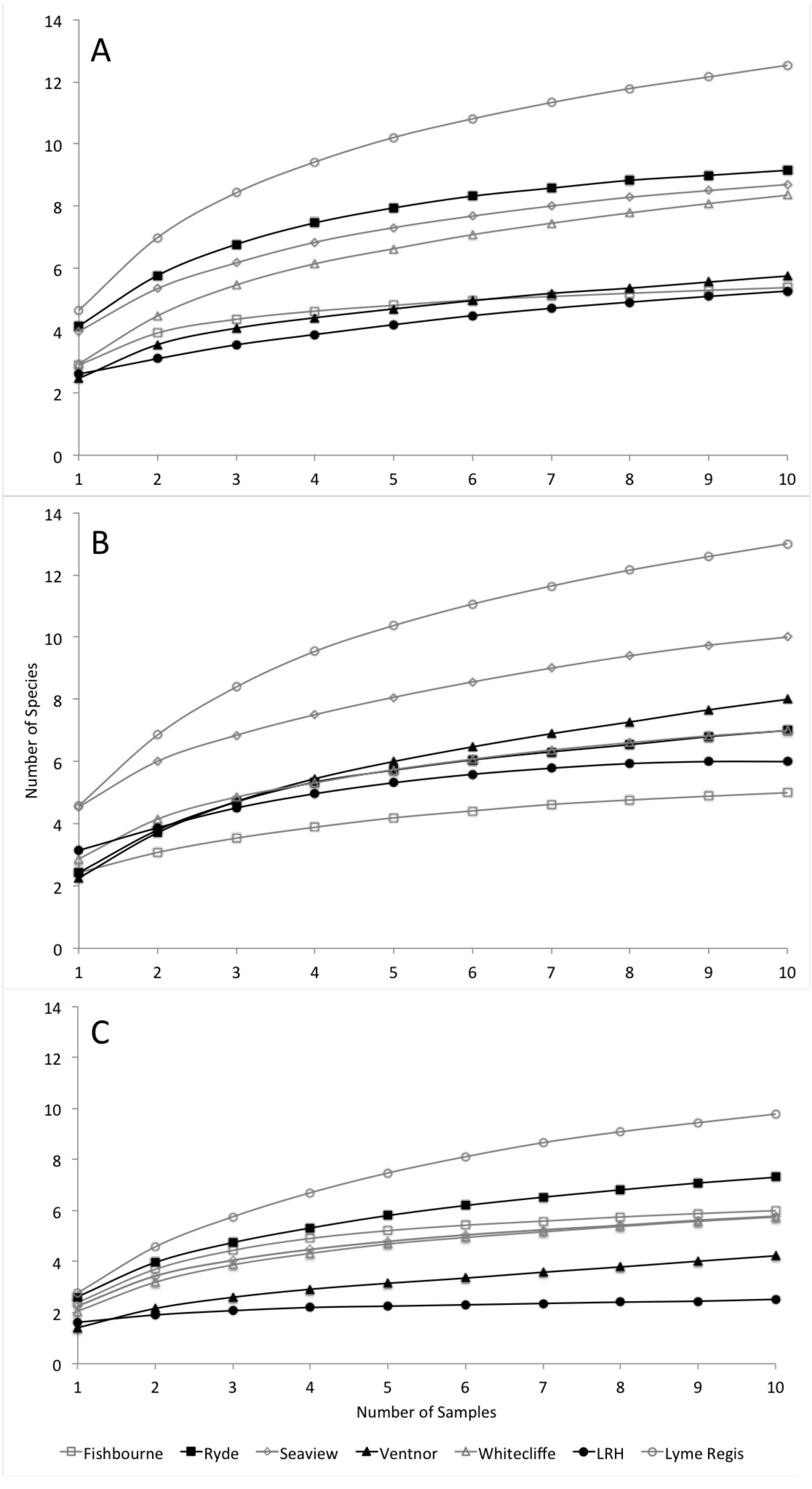

Figure 2.4: Species accumulation curves based on 999 randomisations. A. 10 m spatial scale, B. 1 m spatial scale, C. 0.1 m spatial scale. Natural shores shown in grey and artificial shores in black.

Multivariate non-metric MDS plots of the community data across the seven sites were defined by the factors for site, shore type and spatial scale. For the data plotted using the factor 'sites' and 'shore type' (Figure 2.5) there was some degree of separation between natural and artificial sites, with more clustering of artificial sites toward the top, left of the plot, whereas natural sites were more dispersed with Lyme Regis showing some outliers towards the bottom of the plot. There was, however, still a high amount of overlap between the data points on the plot for both sites and shore types. The MDS plot for spatial scale (Figure 2.6) also demonstrated a high proportion of overlap between the different spatial scales, but there was some indication of separation, where macro scale points were located towards the bottom of the plot, while meso points are very central and micro points are situated towards the top.

Despite the high levels of overlap presented in the MDS plots, further analysis using PERMANOVA showed significant differences for all factors (Shore type: Pseudo-F_1 63.758, $P(\text{perm}) = 0.001$; Spatial scale: Pseudo-F_2 11.38, $P(\text{perm}) = 0.001$ and Site: Pseudo-F_5 28.714, $P(\text{perm}) = 0.001$). Results of the PERMDISP analysis showed homogeneity for factors Shore type and Site, but not for Spatial scale. Although Spatial scale showed significant differences between community data from the PERMANOVA analysis, it had a weaker Pseudo-F value than the other factors and displayed a lack of homogeneity in the PERMDISP analysis. This is likely due to the uneven sample sizes between spatial scale levels, and whilst PERMANOVA is able to analyse data from different sample sizes, it is sensitive to changes between sample sizes and can display significant results due to this rather than actual differences in the data. Therefore, this should be considered an important factor and it is concluded that spatial scale cannot be determined as a significant factor effecting biological communities from these results. For this reason, interactions with spatial scale were also not considered or explored further. However, interactions between Shore type and Site showed significant differences on the biological communities (Pseudo-F_9 3.4746, $P(\text{perm}) = 0.001$). These results suggest that not only are there differences between biological communities on natural and artificial shores, there are significant differences between all sites in general.

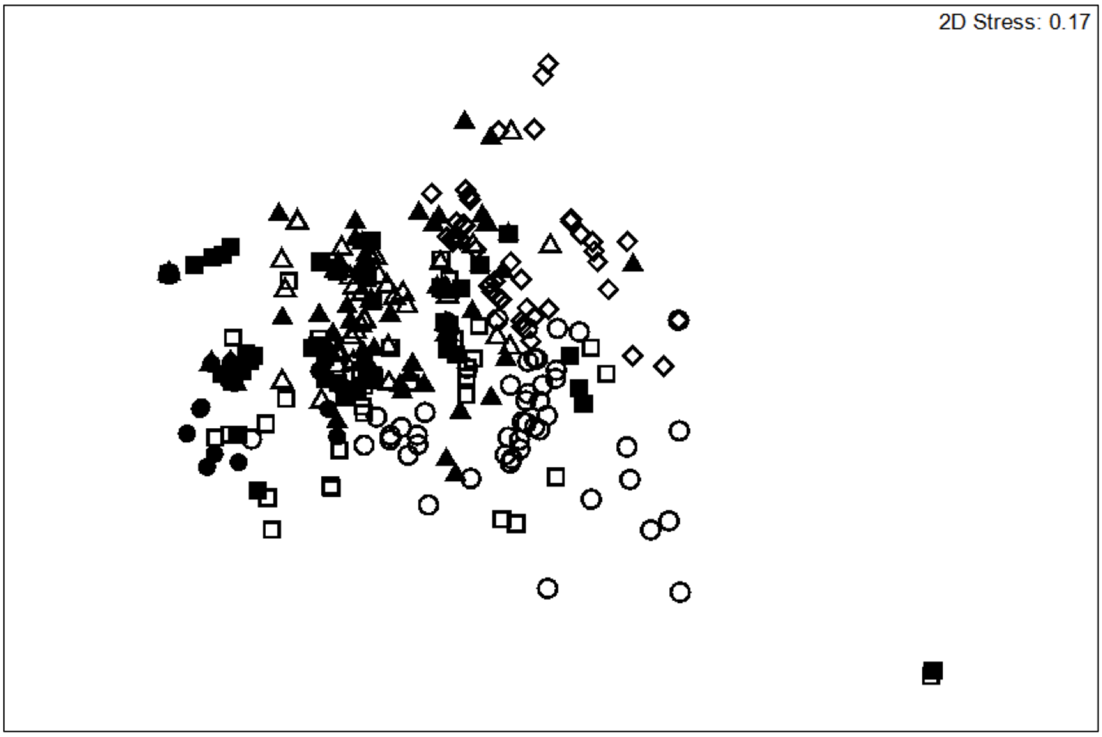

Figure 2.5: Non-metric MDS ordination of Bray-Curtis resemblances between multivariate community compositions for seven coastal sites around the south coast and Isle of Wight, UK. Symbols indicates the seven sites as described in Table 2.4: Lyme Regis (○), Fishbourne (⋄), Whitecliffe (□), Seaview (△), Lyme Regis Harbour (●), Ryde (▲), Ventnor (■)

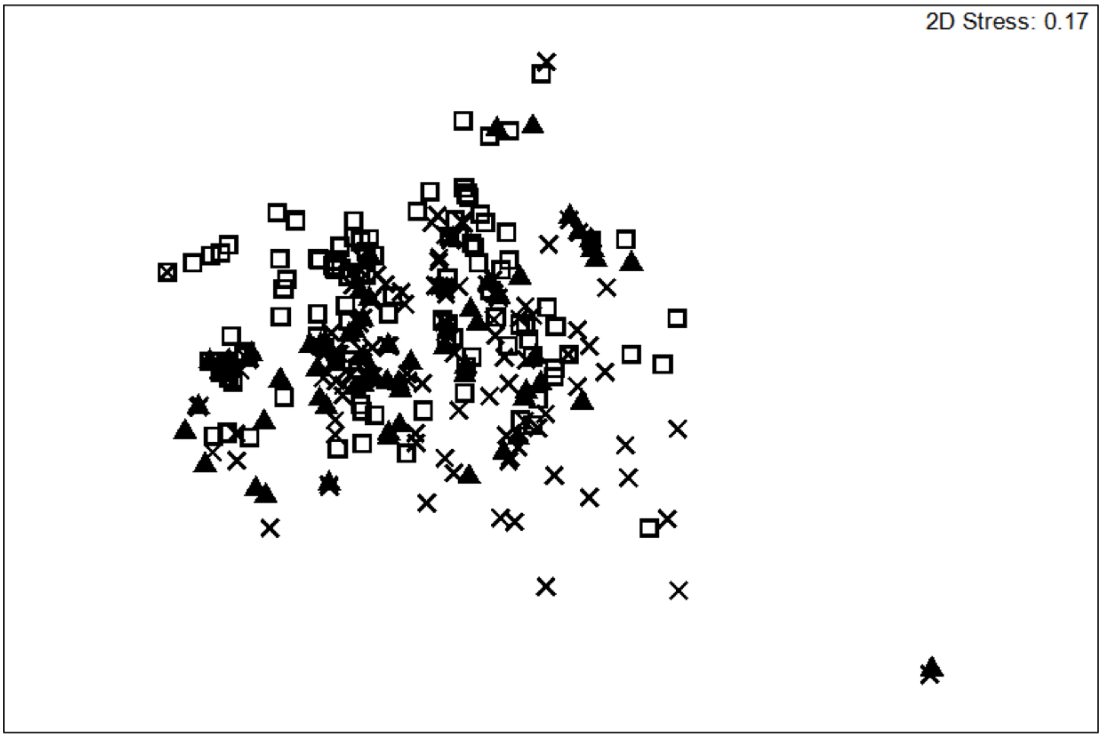

Figure 2.6: Non-metric MDS ordination of Bray-Curtis resemblances between multivariate community compositions for seven coastal sites around the south coast and Isle of Wight, UK. Symbols indicates macro (×), meso (▲) and micro (□) spatial scale as described in Table 2.4

SIMPER analysis identified the key species contributing to the dissimilarity between the communities for the factors Shore type and Sites. Visual inspection of all of the SIMPER results showed the key species which were the most influential species between sites and shore type (Table 2.5). SIMPER results for Shore type show a high level of dissimilarity between communities on natural and artificial shores and highlight that barnacles and *Ulva prolifera* species were the significant contributors to the differences. Site factor overall showed low levels of dissimilarities between sites despite the significant difference presented from the PERMANOVA results. Closer inspection showed that site combinations identified by the SIMPER analysis showed that barnacles and *Ulva prolifera* were also the highest contributing species to the community dissimilarities, however limpets, *Fucus* spp. and *Ascophyllum nodosum* as played roles in differentiating communities between sites. In particular, Fishbourne & Lyme Regis Harbour and Lyme Regis Harbour & Lyme Regis had the highest levels of dissimilarity from all combinations and showed limpets as the highest contributor to dissimilarities followed respectively by barnacles, *U. prolifera* and *F. serratus*. *A. nodosum* was seen to play a 'significant' (>10%) contributing factor between Fishbourne & Lyme Regis Harbour but not for Lyme Regis Harbour & Lyme Regis. For all factors, *Ulva prolifera* was identified as a significant contributor, however, invertebrate species (limpets and barnacles), where shown to be significant, were often seen as the largest contributors to dissimilarities.

Following the analysis of each of the factors, hierarchical clustering using a 1% SIMPROF parameter showed statistically significant cluster groups which were used to form '*supersets*' (Figure 2.7). Collapsed groups within the *supersets* were deemed to have significant internal structures, therefore the *supersets* helped to determine ecologically meaningful groups for further analysis. Further inspection of the data using a non-metric MDS plot with *supersets* groups as a factors (Figure 2.8) showed more separation between groups with some marginal overlap. The MDS groupings also displayed the associated shore type which generally showed one shore type per group, with some mixed. Table 2.6 displays the average similarity for the *Superset* groups A to H, characterising species (along with percentage contribution) and the general community structure in each group. Observations of the community structures and species contributions showed trends in the species colonising groups with mainly natural or artificial, or a mix of associated shores (similar to the visual observation in Figure 2.8). Groups A, E, G and H showed all (or most) samples associated to natural shores, while group D contained samples from artificial shores. The other groups (B, C and F) showed a relatively even mix of samples from both natural and artificial shores. SIMPER analysis of the *Superset* groups (Figure 2.8) showed the key species contributing to the dissimilarity of the group communities.

Table 2.5: SIMPER results for shore type and sites identified the top key species contributing to community differences. *Contribution values provided for species contributing >10%. Only values where dissimilarity divided by standard deviation of contributions (Diss/SD) is >1.

Factor	Interaction	Ave Dissim.	Species - % Contributions*					
			Limpets	Barnacles	*U. prolifera*	*F. serratus*	*F. spiralis*	*A. nodosum*
Shore type		76.50		15.53	15.45			
Site		**0.41**						
	Fishbourne, Ryde	71.06			15.78	12.68		13.73
	Fishbourne, Seaview	74.34		19.32	15.90	10.96	10.45	12.35
	Fishbourne, Ventnor	85.35		19.78	17.14			13.31
	Fishbourne, Whitecliffe	82.43	18.27		15.96		15.43	13.36
	Fishbourne, LRH	**98.06**	**21.81**	**16.75**	**16.02**			**12.24**
	Fishbourne, Lyme Regis	78.60			13.06	11.36		10.80
	Ryde, Seaview	62.21		17.53	13.83		11.23	
	Ryde, Ventnor	69.65		18.20	15.37			
	Ryde, Whitecliffe	68.33	17.73		14.30		13.80	
	Ryde, LRH	77.81	18.99	15.21	18.45			
	Ryde, Lyme Regis	78.69			9.46	10.67		
	Seaview, Ventnor	63.77		18.95	18.25		12.90	
	Seaview, Whitecliffe	67.07	17.71	19.30	15.80		13.88	
	Seaview, LRH	71.94	20.20	15.78	20.71		11.03	
	Seaview, Lyme Regis	81.62		13.00	10.32	10.07	7.61	
	Ventnor, Whitecliffe	71.54	20.38	21.05	17.71		17.71	
	Ventnor, LRH	62.26	28.70	20.88	24.49			
	Ventnor, Lyme Regis	81.68		14.55	12.06	11.79		
	Whitecliffe, LRH	75.09		19.51	18.10		17.50	
	Whitecliffe, Lyme Regis	82.27	13.50		10.96	10.99	11.59	
	LRH, Lyme Regis	**87.87**	**16.07**	**11.11**	**12.58**	**11.50**		

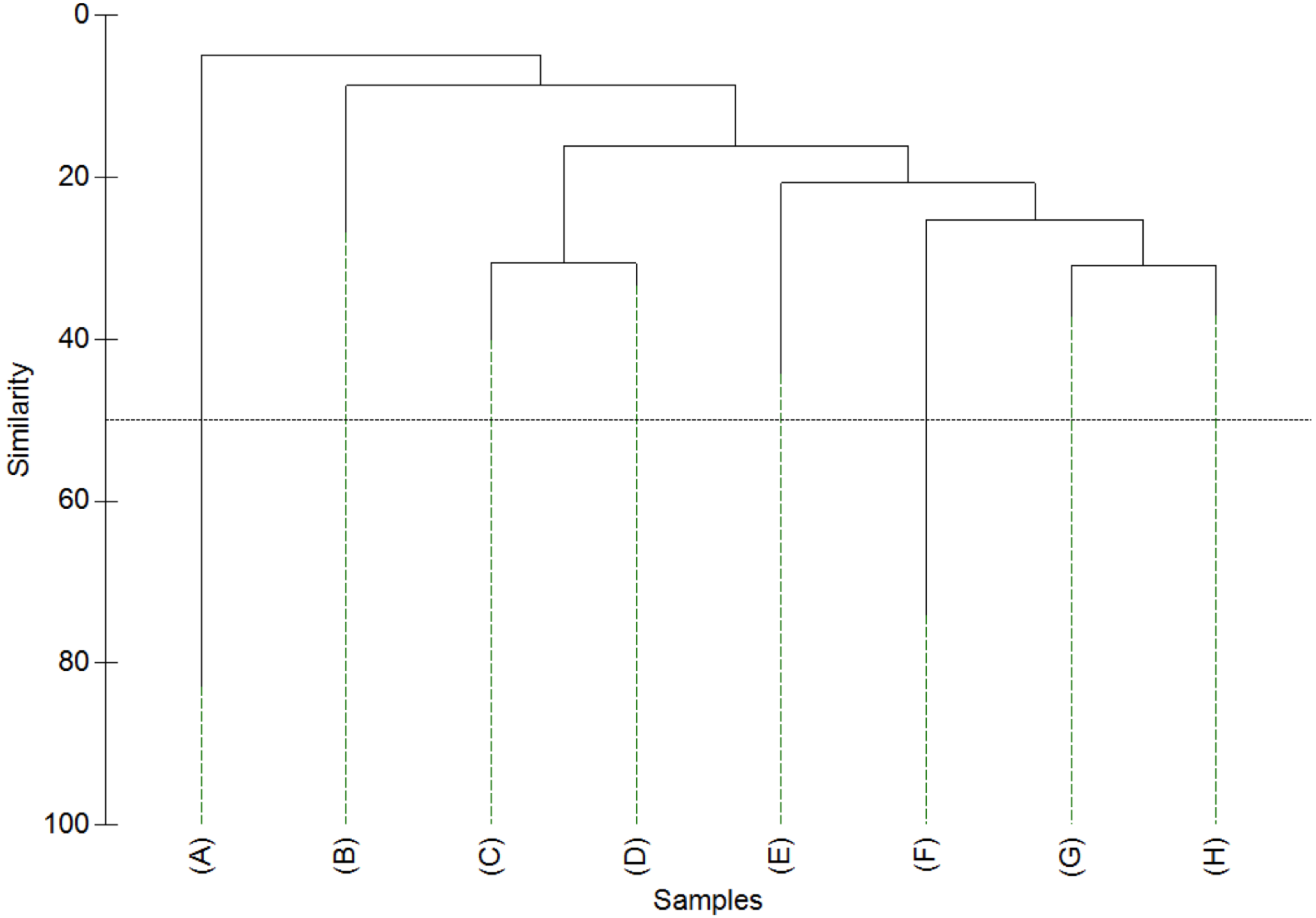

Figure 2.7: Natural and artificial sites from the south coast and Isle of Wight, UK, clustered by group average linkage of Bray-Curtis similarity resemblances between multivariate community compositions and with a 1% SIMPROF parameter to determine superset groups. See Table 2.6 for species within each superset group.

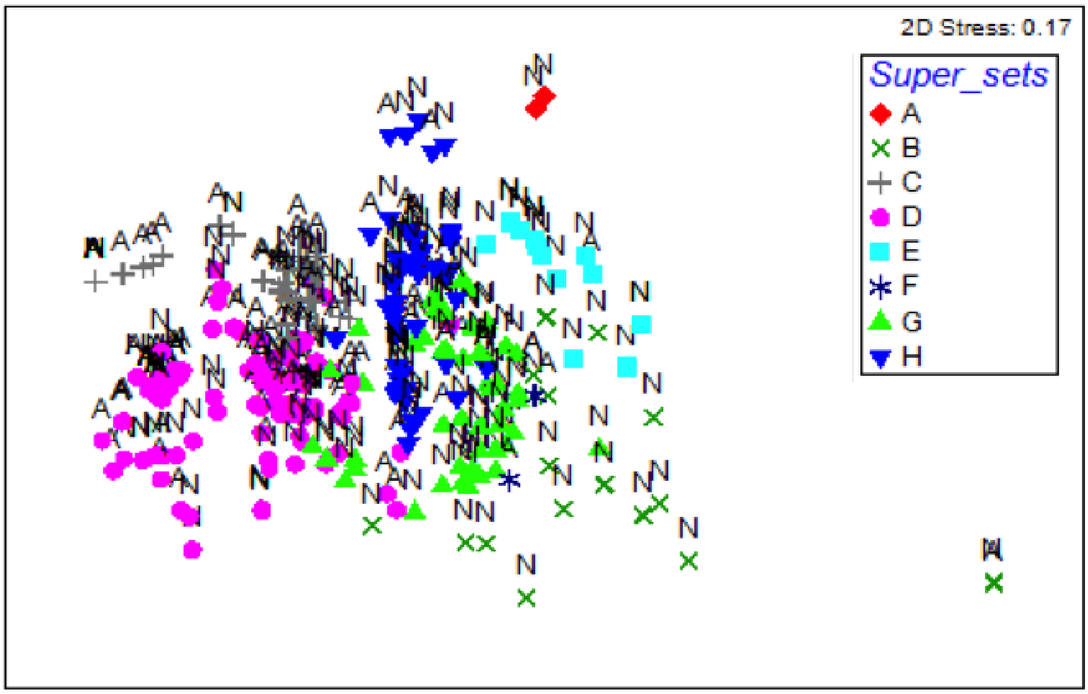

Figure 2.8: MDS of biological data displayed using Supersets generated by 1% SIMPER analysis

Community structure information (Table 2.6), showed characterising species in natural and artificial shores, where fucoids (particularly *F. vesiculosus*: average similarity = 82.84, contribution = 100%) and red algae species (e.g. *A. nodosum*) were associated mainly with natural shores (also present in groups with mixed shore type) and invertebrate species, principally Limpets and Barnacles, were contributing largely to artificial shores. Only *Ulva* spp. were found to be key contributors to both natural and artificial shores but showed, in general, higher average similarities and % contributions to groups associated to natural shores. This reflects theSIMPER results (Table 2.5) which highlight Limpets, Barnacles, *U. prolifera*, *F. serratus*, *F. spiralis* and *A. nodosum* as key contributing species between factor dissimilarities, particularly site.

Table 2.6: SIMPER results showing the average similarity of each *superset* group, the key contributing species within each group and the shore types associated to each *superset* group.

Cluster Group	Group Ave. Similarity	Species	Contribution %	Community Structure
A	82.84	*Fucus vesiculosus*	100.00	All samples from natural shores and contain brown seaweed species.
B	37.27	*Ulva lactuca*	79.31	Samples predominantly from natural shores and consist of algae species.
		Porphyra spp.	5.59	
		Fucus spiralis	4.54	
		Polysiphonia spp.	3.01	
C	54.24	Barnacles	77.63	Samples are from mixed shores and consist of invertebrate and algae species.
		Ulva prolifera	18.24	
D	52.37	Limpets	63.66	Samples predominantly from artificial shores and consist of mainly invertebrate species.
		Barnacles	22.10	
		Ulva prolifera	8.81	
E	57.04	*Fucus serratus*	68.86	All samples from natural shores and contain algae species.
		Ascophyllum nodosum	24.24	
F	74.08	*Chondrus crispus*	71.01	Samples are from mixed shores and consist of algae species.
		Ulva prolifera	28.99	
G	45.71	*Ulva prolifera*,	50.76	All samples from natural shores and contain algae species.
		Fucus serratus	25.92	
		Polysiphonia spp.	11.93	
		Porphyra spp.	3.63	
H	49.84	*Ulva prolifera*	65.02	Samples predominantly from natural shores and consist only of algae species.
		Ascophyllum nodosum	16.37	
		Fucus spiralis	10.97	

Table 2.8: SIMPER results showing the key species contributing to each *superset* group and their average similarity within that group. Only values where dissimilarity divided by standard deviation of contributions (Diss/SD) is >1.

Groups		*Fucus serratus*	*Fucus spiralis*	*Fucus vesiculosus*	*Ascophyllum nodosum*	*Condrus crispus*	*Polysiphonia* spp.	*Porphyra* spp.	*Ulva lactuca*	*Ulva prolifera*	Barnacles	Limpets
A	Av.Sim			82.84								
	Cont.%			100								
E	Av.Sim	39.28			13.83							
	Cont.%	68.86			24.24							
G	Av.Sim	11.85					5.45	1.66		23.2		
	Cont.%	25.92					11.93	3.63		50.76		
H	Av.Sim		5.47		8.16					32.41		
	Cont.%		10.97		16.37					65.02		
B	Av.Sim		1.69				1.12	2.08	29.56			
	Cont.%		4.54				3.01	5.59	79.31			
C	Av.Sim									9.9	42.11	
	Cont.%									18.24	77.63	
F	Av.Sim					52.6				21.47		
	Cont.%					71.01				28.99		
D	Av.Sim									4.61	11.57	33.34
	Cont.%									8.81	22.1	63.66

2.3.2 Topographic Sampling

Ratio values of the topographic complexity were calculated at each level to compare and determine differences between surface roughness at a scale. Average values of the topographic complexity (Surface Rugosity Index) were presented to visualise any clear differences between the spatial scale (Figure 2.9). The Surface Rugosity Index appeared to vary mostly between sites at the higher scales than at 0.01 m where it is relatively the same, however, ANOVA analysis showed no significant differences due to spatial scale (F_2 2.518, p >0.05) or shore types (F_1 3.122, p >0.05). Comparison of the biological data alongside the topographic data showed no obvious pattern between the data sets (Figure 2.10). Furthermore, Spearman's (ρ) correlation analysis (calculated using RELATE analysis) directly compared the biological data and topographic data (for 1.0 m and 0.1 m) and also showed no significant relationship between the datasets (ρ = -0.002, p = 0.54).

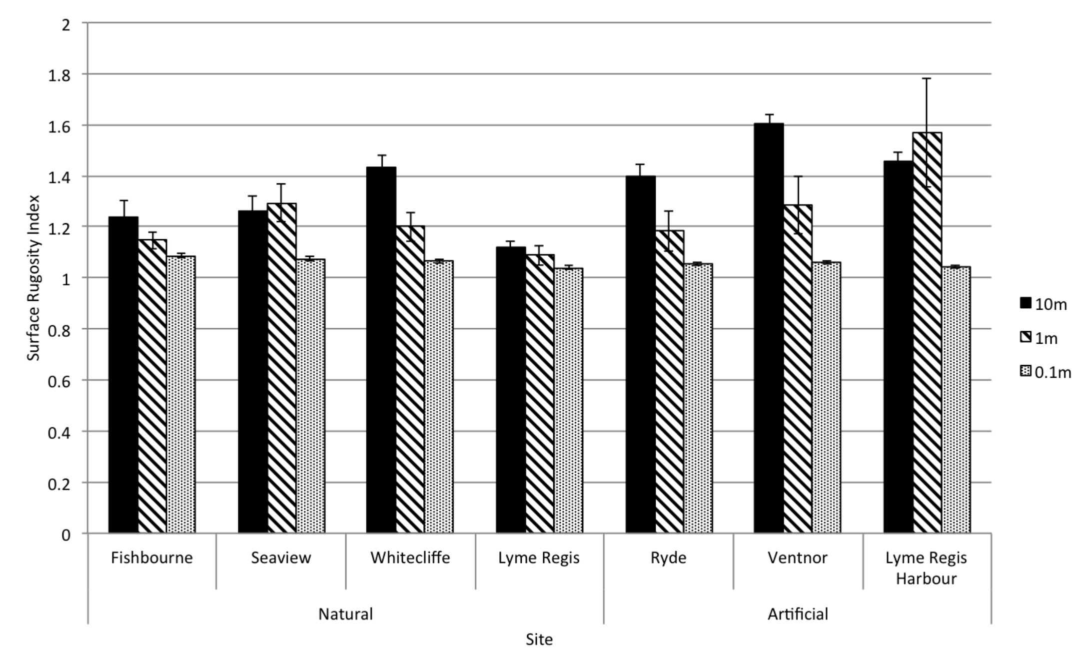

Figure 2.9: Average calculated Surface Rugosity Index for each site at different spatial scale with standard error (SE) bars.

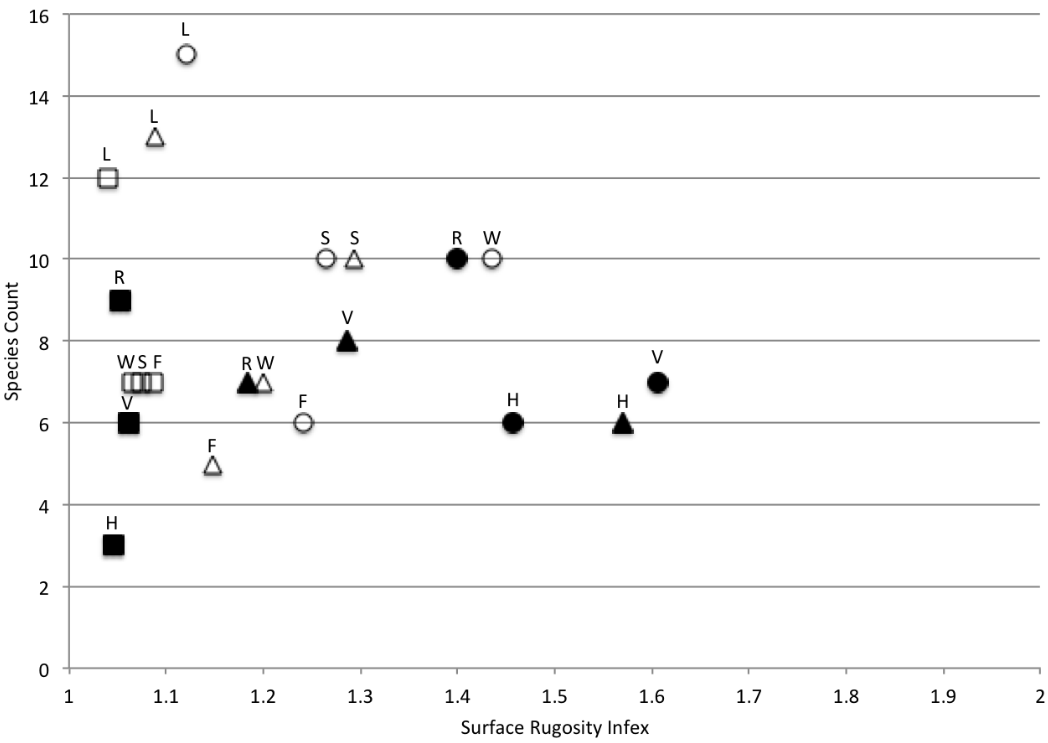

Figure 2.10: Graphic presentation of total species count against Surface Rugosity Index at spatial scale and for shore type where symbols indicates 10 m natural (○) or artificial (●) shore, 1.0 m natural (△) or artificial (▲) shore, and 0.01 m natural (□) or artificial (■) shore. Labels show associated sites: Fishbourne (F), Seaview (S), Whitecliffe (W), Lyme Regis (L), Ryde (R), Ventnor (V), Lyme Regis Harbour (H).

2.4 Discussion

The settlement and development of biological coastal communities is determined by a number of natural abiotic and biotic factors (Raffaelli and Hawkins, 1996) and increasingly anthropogenic impacts (Benedetti-Cecchi, 2000; Airoldi and Bulleri, 2011). In this study we explored the biological communities and topographic differences between seven intertidal natural and artificial shores. In addition, we investigated whether variability on different scales plays a significant role in differences in the settlement and colonisation of natural shores and artificial hard structures.

2.4.1 Comparison of natural and artificial shores

The ecological communities as a result of the shore type factors explored in this study showed that there was significant difference between species assemblages on the natural and artificial shores studied. In particular, six key species were identified as community differentiators between shore types and sites (Table 2.5). Further analysis of the results by defining ecologically coherent groups (Figure 2.7) showed that shore type may determine the characteristic species that settle on either natural or artificial shores. My study showed that natural shores tend to be colonised by species such as fucoids and some foliose red algae (strong % contributions), while artificial shore communities are largely determined by invertebrate species such as limpets and barnacles (Table 2.8). This distinction between natural and artificial communities is likely due to the placement of artificial shores (CDS) in areas of higher environmental stresses (Moschella et al., 2005; Burcharth et al., 2007; Vaselli et al., 2008; Pister, 2009; Firth et al., 2013b), where only more adaptable and hardy species can withstand the stronger physical conditions or are able retreat to sheltered areas will survive (mobile fauna). Invertebrate species (in general) have higher mobility, enabling them to choose suitable habitats with lower environmental pressures and competition, or move when under threat. On the other hand, algal species are sessile organisms and must withstand the environmental pressures in the location they settle in, or die. Therefore, on artificial shores with higher wave exposure and physical stresses, opportunistic species such as *Ulva* spp. thrive and tend to dominate surfaces (Bunker et al., 2010; Maggs et al., 2007).

My study also aimed to examine the relationship between communities and topographic complexity. Following previous research where structural complexity is identified as one of the key differences between natural and artificial shores (Chapman, 2003; Moschella et al., 2005; Aguilera et al., 2014; Evans et al., 2016; Hawkins et al., 2016) it was hypothesised that natural shores will have a higher topographic complexity than artificial shores, and will therefore have a great species diversity. In particular Burt et al. (2009b), Coombes et al. (2011) and Firth et al. (2012) showed that rougher surfaces account for increased biodiversity, compared to smooth and harder materials. Furthermore, previous studies claim that spatial scale plays an important role in habitat availability, and could also be a determining factor between natural and artificial shores (Moschella et al., 2005; Burt et al., 2009b; Coombes et al., 2011; Firth et al., 2012, 2014; Evans et al., 2016). My study, however, found no significant relationship between topographic complexity (surface rugosity) and biological communities. In addition, despite displaying a significant difference between communities at difference spacial scales, it was determined that this difference could be in fact due to unbalancing in the experimental design between spatial scales, and not due to actual difference in the data. However, this is not conclusive and therefore I was unable to fully determine whether spatial scale does in fact play a significant role between biological community compositions.

Site as a factor, displayed a significant difference between biological communities through the PERMANOVA analysis, but showed lovel dissimilarity in the SIMPER results. Furthermore, there was a significant difference in biological communities due to shore type and site as interacting factors. Species richness varied across all of the sites, but interestingly, Lyme Regis and Lyme Regis harbour (neighbouring sites with the same environmental conditions, but very different topography with a flat natural shore and much steeper breakwater) recorded the highest and lowest species richness respectively (Figure 2.3). Further inspection of the communities between site combinations also reflected that Lyme Regis and Lyme Regis harbour had the highest levels of dissimilarity between the biological communities (Table 2.5). The overall results for the factor 'Site' and the within factor comparisons indicate that biological communities recorded across all seven site location vary greatly, and are likely highly different to each other independent of shore type. Differences are more likely due to other interacting environmental and physical stresses (Benedetti-Cecchi, 2000; Airoldi and Bulleri, 2011; Evans, 2016). This observed difference is likely due to a combination of both functional diversity, where variation is due to the individual species in the communities and their individual functions and ability to survive different stresses, and β diversity (between-habitat diversity) (Airoldi et al., 2008; Dornelas et al., 2014; Magurran, 2016). Table 2.4 shows the physical differences between each site, and even within natural or artificial categories, sites vary in terms of their wave exposure, surrounding habitats, position on the shores and materials. Thus, species successfully colonising each site will be adapted to accommodate the physical conditions of each site.

2.4.2 Factors determining community structure

There was significant results supporting the differences between biological communities on all of the shores. Although it cannot be said that there was a distinct difference between species diversity on natural and artificial shores, this study shows that shore type does determine community characteristics, therefore developing different assemblages. This is likely for reasons other than just 'shore type', and could be due to a number of interacting factors such as wave exposure, slope, substrate material and inundation levels (Benedetti-Cecchi, 2000; Evans, 2016). Artificial sites were observed to have higher wave exposure compared to natural shores, and to be located within the lower regions of the shoreline (sub-littoral fringe to eulittoral shore). These conditions can modify habitat type and environmental stresses which can lead to risks of desiccation, dislodgement from the shore, competition for optimal habitat space and exposure to predators and lack of food resources (Haslett, 2000; Kendall et al., 2004; Jackson and McIlvenny, 2011). It was also noted that (in general) natural shores appeared to have higher abundance of micro-habitats than artificial shores. Micro-habitats such as cracks, crevices and rock pools, provide essential refuge spaces to protect species during low tide or strong wave conditions (Chapman and Blockley, 2009; Carvalho et al., 2013), and generally lacking on artificial shores compared to natural shores (Chapman, 2003; Moschella et al., 2005; Aguilera et al., 2014; Evans et al., 2016; Hawkins et al., 2016). With that said, artificial shores demonstrated habitat availability and refuge spaces on a different spatial scale. There was a clear lack of micro-habitat heterogeneity, however, there was more large scale complexity formed through artificial structural designs compared to the natural shores. Artificial shores are often constructed using large rock units, and therefore create a modular structure. This structural formation differs to natural shores which have no modularity and lack larger scale areas within the shores. Although the complexity level could be considered lower and less favourable to intertidal assemblages, it has been shown to support a number of species, particularly invertebrates, and may provide 'hidden' internal habitats spaces between units which are inaccessible to record (Firth et al., 2012). With this in mind, it would be worth quantitatively assessing how microhabitats vary in general amongst sites, both natural and artificial, and whether this is more likely a significant factor diversifying communities. Another potential factor that could affect the communities between sites is the substrate material. Previous research by Coombes et al. (2011) shows that hard materials often used in coastal engineering due to their high strength and durability properties, are less suitable for increasing biodiversity. This study showed a variation in different material substrate types between sites which could contribute to any differences.

In summary, this study showed no significant differences between species richness and topography between sites or spatial scale. Furthermore, topography did not show any correlation with biological diversity, therefore there we must reject all of the hypothesis.

2.5 Conclusions

Overall this study showed significant difference between biological communities on natural and artificial shores, in particular characterising species within the communities. Natural shores were colonised by species such as fucoids and some soft red algae, while artificial shore communities were largely determined by invertebrate species such as limpets and barnacles. Furthermore, this study also identified no significant difference relationship between topographic complexity and biological communities. Observed differences between micro and macro habitats on natural and artificial shores indicates structural complexity formed between the rock unit spaces could provide potential habitats to support intertidal species. Access to these areas limited sampling in these areas and therefore it was not possible to determine their role as habitat spaces.

Chapter 3

Hidden biodiversity in cryptic habitats provided by porous coastal defence structures

Abstract

In response to flood risk from rising and stormier seas, increasing amounts of natural coastline worldwide are being replaced by a proliferation of coastal defence structures. While the primary role of defence structures is protecting the coastline, consideration should be given to the biological coastal communities they support. Artificial structures are currently seen as poor habitats for marine organisms. They are constructed in harsh coastal environments, lack structural complexity, and are subjected to episodic disturbance from maintenance, making them undesirable habitats for coastal species. Recent work has focused on mitigating the impacts of coastal defence structures, through secondary routes such as enhancing biodiversity by encouraging colonisation of marine biota. Research thus far has focused on enhancements to improve structural complexity on the external surfaces of coastal defences. Many structures are porous with internal compartments. To date no work has been undertaken on the habitat provided by the internal surfaces of the blocks used in building structures.

I investigated the role of porous coastal defence structures in habitat provision. Taking advantage of a groyne reduction from 45 m to 20 m length, I surveyed the internal environment of the structure. I also considered the impacts of maintenance activity on coastal assemblages. My work shows that the internal environment of artificial structures provides functional habitat space supporting higher species richness and diversity than external surfaces. The more benign environment of internal surfaces protects from desiccation stress and probably experiences less scouring by mobile sediments, and as

such is of unrealised importance as a habitat within coastal defence structures. External surfaces are also subject to higher levels of biotic and anthropogenic disturbance such as wave exposure and maintenance activities, making them less desirable artificial habitats. These findings reveal the multifunctional role of porous coastal defence structures, acting as engineering protection and habitats for coastal assemblages.

3.1 Introduction

Coastal areas provide essential economic resources and satisfy a variety of societal needs. Coastal ecosystems account for a substantial proportion of global ecosystem services (Costanza et al., 1999; Martínez et al., 2007), including coastal protection (Garcia et al., 2004; Bulleri, 2005; Chapman and Underwood, 2011; Dugan et al., 2011). Faced with the effects of accelerated climate change, coastal regions are susceptible to flooding and loss of land, requiring adaptational actions (Nicholls and Mimura, 1998; Airoldi et al., 2005; Burcharth et al., 2007; Philippart et al., 2011). The development of coastal defence structures (CDS) is fundamental in protecting land, property, infrastructure and other economic resources. Thus, in many areas worldwide, coastlines are dominated by artificial structures (MAFF, 2000; Airoldi et al., 2005; Bulleri and Airoldi, 2005; Moschella et al., 2005; Firth et al., 2013b; Liquete et al., 2013; Firth et al., 2014) causing significant changes to shores through loss, replacement or fragmentation of natural habitats. This places intense pressure on coastal resources and the environment, and affects the structure and function of related marine ecosystems (Connell and Glasby, 1999; Bulleri and Chapman, 2004; Airoldi et al., 2005; Airoldi and Beck, 2007).

Infrastructure placed in any natural environment will inevitably become colonised by primary settlers such as epibenthic marine organisms and biofoulers (Evans et al., 2016). Artificial structures can be viewed as surrogate habitats for natural shores (Connell and Glasby, 1999; Moschella et al., 2005; Burt et al., 2011). With the aid of additional structural modifications to increase habitat heterogeneity, increased colonisation and enhanced biodiversity of marine species on artificial substrates can be encouraged (Firth et al., 2013b,c; Evans et al., 2016). Currently, CDS are seen as poor substitutes for natural rocky shores because they support lower species diversity (Bulleri and Airoldi, 2005; Bulleri et al., 2005; Moschella et al., 2005; Chapman and Blockley, 2009). Coastal defence structures are typically built in high-energy environments with stronger wave action than most natural rocky shores (Jonsson et al., 2006; Burt et al., 2011; Evans et al., 2016), providing harsh habitat conditions for common rocky shore organisms, and opportunities for invasive species through new hard substrata (Airoldi and Bulleri, 2011; Firth et al., 2013a). These conditions are made worse by scouring from sand, gravel and cobbles (Moschella et al., 2005; Bulleri and Chapman, 2010). Coastal defence structures are also less topographically complex than natural rocky shores (but see Chapter 2), reducing habitat and microhabitat provision (Martins et al., 2010; Hawkins, 2012). Their extent is often smaller than natural shores (Moschella et al., 2005), inevitably leading to a restricted species pool and altered biological interactions amongst species (Bulleri, 2005; Bulleri et al., 2005; Jackson et al., 2008; Bulleri and Chapman, 2010; Coombes et al., 2015).

In conjunction with factors considered above, there is constant pressure on structural integrity of coastal defences due to erosion, scouring, overtopping and undermining (Kamphuis, 2010; Airoldi and Bulleri, 2011; Firth et al., 2013b). Over time this can affect the stability and function of the structure, requiring maintenance (Dayton, 1971; Sousa, 1979; Airoldi, 2003; Moschella et al., 2005). Maintenance, however, can result in severe ecological disturbance. It can remove large areas of the habitat and causes disruption to settled communities by the removal and replacement of part or all of the structures (Tsinker, 2004; Airoldi and Bulleri, 2011). Such works can dislodge, crush or expose colonising species, potentially reducing biodiversity and opening up space to opportunistic species (Dayton, 1971; Sousa, 1979; Hutchinson and Williams, 2003). Large costs are also incurred in the upkeep of the structures (Roebeling et al., 2011).

Porous rock defence structures are widely used in coastal engineering (Crossman et al., 2003). They serve a practical role in the protection of our coastline by reducing wave transmission, reflecting incident waves from the shores, and dissipating wave energy (Dalrymple et al., 1991; Losada et al., 1995; Garcia et al., 2004; Burcharth et al., 2015). Wave dampening is an important function that many other impermeable defence structures do not provide sufficiently (Garcia et al., 2004). The porous structure allows some of the waves energy to pass through whilst creating flow resistance and some reflection from the structure, resulting in turbulence through the porous medium and dissipation of wave energy (Silva et al., 2000; Garcia et al., 2004; Jung et al., 2012). Consequently, essential protection to the shoreline is provided whilst still allowing the natural process of water run-up on the coast. This imitates many natural shoreline barriers, such as coral reefs, mangroves and rocky shores, which can provide natural protection against waves and storm surges (Lowe, 2005a,b; Monismith, 2007; Fernando et al., 2008; Hu et al., 2014).

Porous defence structures, compared to solid structures such as sea walls and revetments, are also seen to be more environmentally friendly because they have a smaller physical footprint creating less disturbance to benthic soft sediment organisms (Koraim and Rageh, 2013), and can be more aesthetically pleasing (Garcia et al., 2004). Considerable recent work has focused on improving secondary functions of coastal defence structures, particularly enhancing their colonisation by marine biota. Research into artificial enhancements such as boring holes to create rock pools and drilled grooves to increase heterogeneity have been extensively researched (Moschella et al., 2005; Chapman and Blockley, 2009; Borsje et al., 2011; Naylor et al., 2011; Firth et al., 2012, 2013b,c, 2014; Coombes et al., 2015; Evans et al., 2016). Other studies have investigated the use of different materials to encourage settlement on the surface of these structures (Coombes et al., 2011; Coombes, 2011; Green et al., 2012; Coombes et al., 2013). Whilst this work has been a successful and an integral step towards working with nature by creating "green" infrastructure, the focus has been solely on the external surfaces of sea defence structures. To date no work has been undertaken on the habitat provided by the internal surfaces of the blocks used in building porous defence structures because of logistic constraints. Thus, this study presents the first opportunity to document the internal section of a porous rock armour structure. This is potentially a habitat providing some refuge from the harsh physical conditions of the intertidal zone in general (e.g. desiccation and wave action) and defence structures in particular (e.g. scouring).

The use of porous structures in coastal engineering can be viewed as providing a multifunctional role, protecting vulnerable coastlines and supporting intertidal communities. This chapter compares biodiversity of internal versus external surfaces taking advantage of the reduction of a groyne from 45 metres to 20 metre extent at Highcliffe on the South coast of the UK as part of a reconfiguration of an existing coastal defence scheme. This is the first time the internal section of a porous rock armour structure has been studied. We compare the community composition, abundance and diversity of species in different habitats upon and within the various habitats sheltered by a coastal defence structure. In addition, we evaluate the extent of anthropogenic disturbance caused by the removal process, to indicate potential levels of general coastal defence maintenance disturbance.

3.1.1 Aims and Objectives

In this chapter the community composition, abundance and biodiversity of species recorded upon and within the various habitats offered by a porous CDS, are compared. More formally the following hypotheses are tested:

1. internal habitats on the porous defence structure will support greater abundance and diversity of species than external habitats;
2. internal habitats will support a higher number of invertebrate species than external habitats.

3.2 Methodology

3.2.1 Study Site

The study took place at Highcliffe in Christchurch Bay on the south coast of England, UK (Figure 3.1). Christchurch Bay has a steadily eroding coastline of Barton and Headon clay beds and tertiary deposits exposed as cliffs. It experiences a low amplitude double high tide, which is characteristic of the Solent area. In spring tides the area experiences fluctuations in mean water levels of approximately 1 m. There is also a complex tidal current system that circulates within the bay and a south-westerly wave pattern causing high-energy beaches to the west and local sediment drift and erosion. The area receives some protection from the Isle of Wight situated to the east and Durlston Head to the West (Tyhurst, 1986). The Highcliffe coastal defence scheme regularly undergoes maintenance works due to the high energy conditions, therefore Christchurch Borough Council (CBC) decided to revert from timber to rock groynes in 1992 (making the groynes over 20 years old at the time of sampling) to increase strength and durability. The defence currently comprises eleven rubble mound groynes, consisting of short and long structures (30-45m) and a bastion, made from Portland Oolitic limestone (Tyhurst, 1986; Harlow, 2013) (Figure 3.2). In addition, the coastal defence scheme has a sea wall structure for protection against extreme high tides, but rarely under undergoes inundation. The groynes are designed with 1 in 2 side slopes, 1 in 2.5 roundhead slopes and a 4 m crest width (Harlow, 2013). These are situated amongst a mixture of shingle and sand beaches (CBC, 2008), and the structures are estimated to sit approximately 1 m into the substrate. The porosity of the structures was unknown. In 2012, CBC deemed the groyne system at Highcliffe to be over engineered with a number of the groynes not being fully utilised within the coastal defence system. Therefore it was decided that the best approach was to remove and recycle the rock units. This area regularly undergoes routine maintenance work that consists of the replacement of rock units, removal/ replacement of sand, or in some circumstances the partial reconstruction of a structure. The management of this area is essential to retain the current coastline, protect residential properties and maintain the shoreline for tourism and local amenity use.

3.2.2 Groyne Removal

The groyne reduction took place during the lowest spring tides in June and July 2013 by CBC coastal engineers. The process removed 102 individual rock armour units of varying sizes (1-4 tonne rock units) roughly rectangular in shape, using a digger with a grab or bucket. The size of the structure was reduced from 45 m to approximately 20 m. Surface rock armour units from the end (nose) of the groyne were the first to be removed,

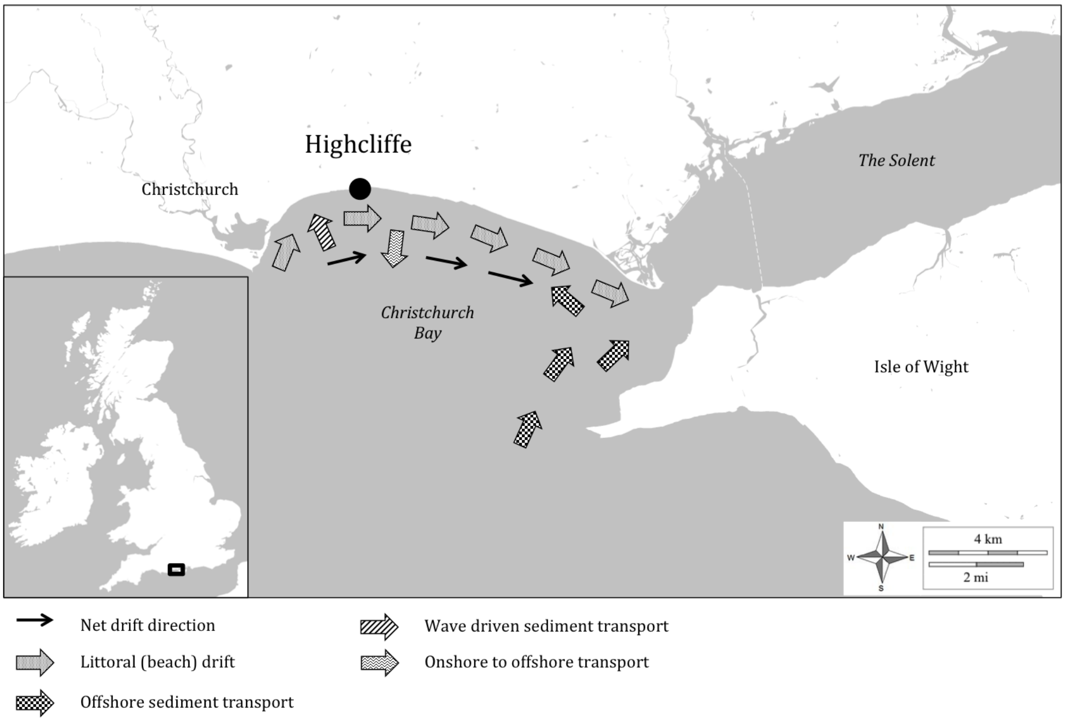

Figure 3.1: Study area selected is Highcliffe situated within Christchurch Bay on the South coast of the UK. Arrows show the sediment transport activity in the bay. Image adapted from MMIV © SCOPAC Marine Inputs map.

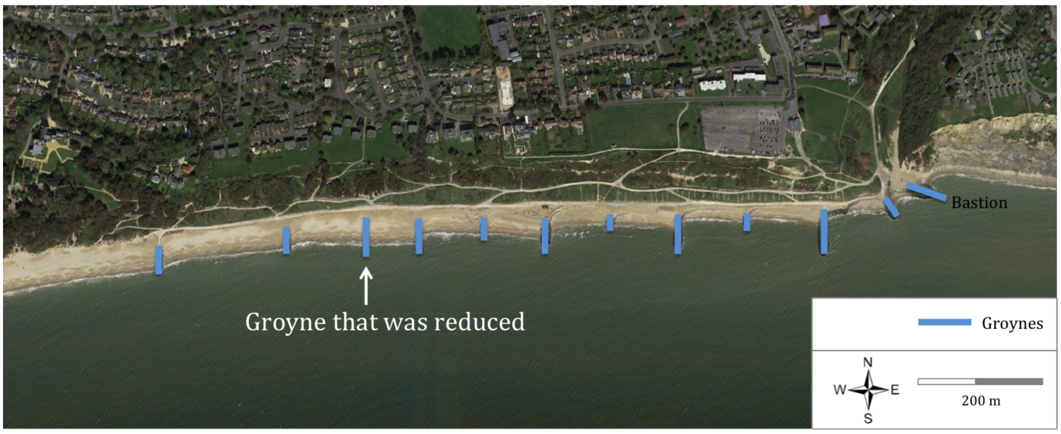

Figure 3.2: Image of the groyne system constructed at Highcliffe within Christchurch bay. Image shows the eleven rock groynes and a bastion of varying long and short lengths, and highlights the study groyne that was reduced. Image adapted from Imagery ©2016 Google, TerraMetrics, Map data ©2016 Google.

exposing the foundation rocks. This allowed access to larger 4 tonne rock armour units that had sunk approximately 1 m into the sediment since they were installed. After the seaward nose and initial foundation units had been removed, the top and side layers were extracted followed by the internal (central) units. Owing to the short tidal window available for work, it was essential for the engineering work to be done in the specific removal order detailed above to ensure that structural integrity was retained between

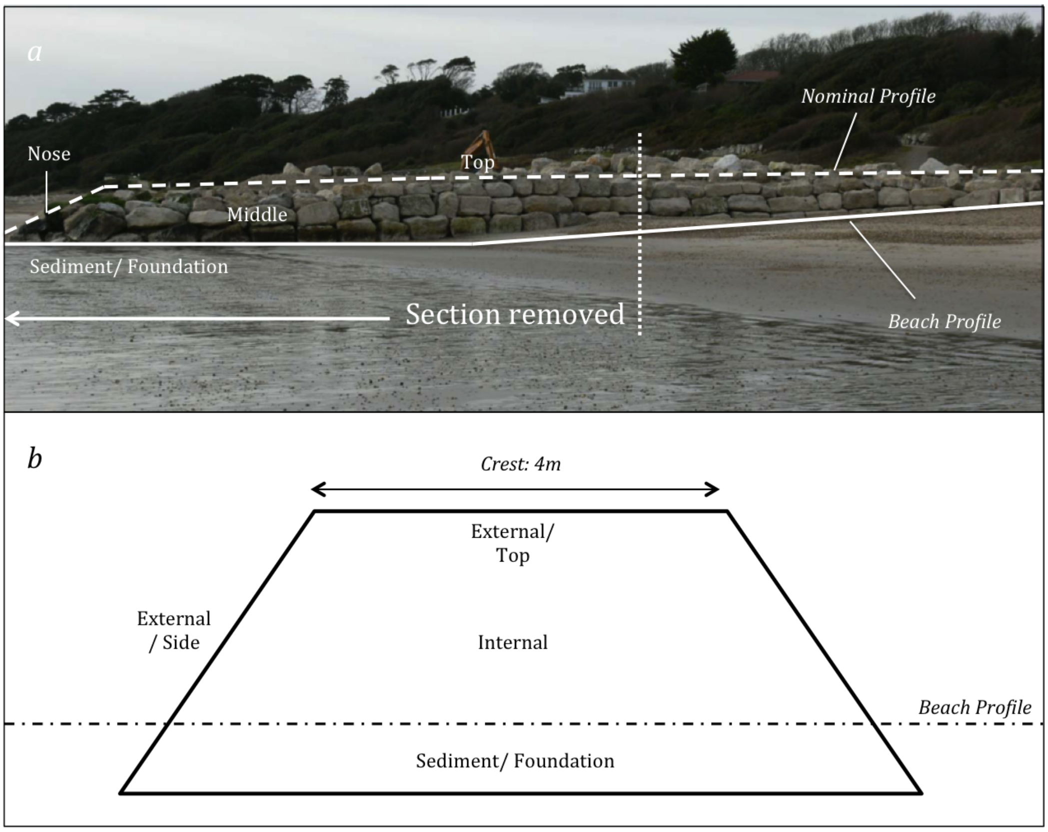

Figure 3.3: Displays the areas categorised within each factor. (a) Shows a side view of the groyne that was removed, and the location of the categories under each factor that are visible. (b) Illustrates a landward facing cross-sectional representation of the groyne, and the relative locations of the categories under each factor.

removal periods.

3.2.3 Data Sampling

The restricted timeframes meant that ecological sampling was carried out around the engineering works; therefore all information was recorded in situ. Photographs and physical details of each unit were recorded, along with the surface area of each unit face and estimated weight of the unit (provided by the on-site CBC engineers). Each unit face was recorded as an individual sampling point, and categorised by three different factors to determine the position and environmental exposure of each unit face (Figure 3.3):

1. exposure to environmental conditions (external - outside unit face with seaward orientation and wave exposed, external sheltered - outside unit face with landward orientation, internal - unit face located within the groyne, sediment - unit face located within soft sediment due sinking over time);

2. elevation on the shore (foundation - lower shore, middle, top of the shore);
3. placement of unit faces within the structure (nose - end of the structure, internal, side, top). Connections to other rock armour units were also noted.

Biological sampling was conducted for each unit face by identifying organisms present to species level where possible with counts for mobile fauna and percentage cover for sessile species. Where it was not possible to identify to species level, identification was made to the closest taxonomic level. Maintenance disturbance was classified as areas of the unit face where fracturing and/or removal of the surface was visible due to the removal process. The level of maintenance disturbance was estimated by calculating the percentage of the unit face damaged or removed. The number of occurrences per rock armour face and frequency of occurrences out of the total sample were also logged.

3.2.4 Statistical Analysis

To test our hypotheses, statistical analyses were carried out using Simpsons Index of Diversity (D), PRIMER-E ver. 6 and PERMANOVA statistical software (Clarke, 1993; Anderson et al., 2008; Clarke et al., 2014) to determine the difference between species diversity, richness and percentage abundance of species recorded on a rock groyne in relation to exposure levels, elevation on the shore, and placement within the structure. There were three fixed factors used in the analysis of the experiment: exposure (four levels: external, external sheltered, internal or sediment), elevation on the shore (three levels: foundation, middle or top), and placement within the structure (four levels: nose, internal, side or top unit).

To test our first hypothesis that internal habitats on the porous defence structure will support greater species diversity than external habitats, species diversity was calculated for each replicate using the Simpsons Index of Diversity (D) (Equation 3.1) (Simpson, 1949; Magurran, 2004). In Equation 3.1, n = the total number of organisms of a particular species and N = the total number of organisms of all species. This indexs orientation towards the more common species gives a more accurate description of species composition. Index values were compared using a one-way ANOVA and Tukeys HSD post-hoc test.

$$D = 1 - (\Sigma n(n-1)/N(N-1)) \tag{3.1}$$

Next, multi-dimensional scaling (MDS) plots based on a Bray-Curtis similarity matrix (Bray and Curtis, 1957) of square-root transformed data were created for each factor (exposure, elevation on the shore and placement within the structure) to visualise patterns using rank similarities and hierarchical clustering in the multivariate output (Clarke, 1993; Clarke et al., 2014). Furthermore, Permutational Multivariate Analysis of Variance (PERMANOVA) was used to test multivariate species assemblages based on 9999 unrestricted random permutations of residuals (Anderson et al., 2008). A three-way factorial design was used with the three factors specified previously. PERMANOVA routine was then used to test differences in the species assemblages specifically in response to exposure levels using within factor pair-wise comparisons (an alternative to running a one-way ANOVA). PERMANOVA was selected for multivariate and univariate analysis as it is more flexible with regards to assumptions than univariate parametric statistics by using permutations to make it distribution free, and also making it more suitable and robust in particular for analysing multivariate species data and data with several zero values (Anderson et al., 2008).

To address our second hypothesis, percentage contributions of individual species to dissimilarity between communities were calculated using the Similarity Percentage (SIMPER) (Clarke, 1993; Anderson et al., 2008; Clarke et al., 2014). Community structure was compared in the 'internal' and 'external' to investigate their role as habitats at different exposure levels. For the maintenance disturbance, we calculated the average disturbance as a percentage of the surface cover, the number of occurrences and the extent of the damage as a percentage of the total number of samples for each factor level, to provide indicative data which may be used to inform methods for reducing disturbance levels. A Chi-squared test was then used to determine if there was significant differences in levels of maintenance disturbance overall and for each factor, based on the assumption that there should be no difference between maintenance disturbance within or between each factor.

3.3 Results

3.3.1 Ecological Sampling

A total of 102 rock units were removed from the groyne structure, and the faces of each unit recorded. Species recorded during the removal process for the exposure, more specifically the presence of species recorded on the internal and external rock unit faces and their percentage cover, are displayed in Table 3.1. Mean percentage cover and Simpson's Diversity Index (D) were calculated across all internal and external unit faces in order to determine the difference between species diversity due to exposure levels. Internal faces supported a higher number of species than external unit faces (internal 20 species, external 10 species) (Table 3.1), particularly for invertebrate species and red seaweed species. Mobile fauna such as *Eulalia viridis*, juvenile *Carcinus maenas* and

Nucella lapillus were all found only on internal faces. There was, however, a higher mean percentage cover of species found on external faces (53% ±50%), than on internal faces (25% ±61%) (Table 3.1).

The results in 3.1 suggest that this is due to the presence of the alga *Ulva* spp., found in both environments, but more abundant on the external faces. *Ulva* spp. was recorded to cover on average 47% (±31%) of external faces compared to 5% (±9%) of internal faces. Overall, despite there been a higher percentage cover on external surfaces compared to internal surface, the difference was deemed insignificant (F_1 0.351, p >0.05). This is likely due to the large variation in mean coverage across all internal and external surfaces. Furthermore, Simpsons Index of Diversity (D) showed overall that internal surfaces had higher species diversity (0.90 ±0.002) than external surfaces (0.82 ±0.006), however this was not significantly different (F_1 0.418, p >0.05).

MDS plots (with 25% similarity contours) were created for each factor to visualise patterns in the community data. Figure 3.4 *a*, an MDS plot for exposure factors, showed clear separation of in exposure levels, particularly internal and external, suggesting differences in the communities recorded in those areas, whereas external sheltered and sediment levels appeared to be more dispersed. This is reflective of the sampling observations for external sheltered and sediment areas which had very few species (if any in the sediment areas) colonising. Figures 3.4 *b* shows no obvious pattern due to elevation on the shore. Foundation level units can be seen to cluster towards the right hand side of the plot, however there is still large scattering and overlapping between all of the data. Figures 3.4 *c* shows no pattern at all in the data due to placement within the structure and large dispersal and overlapping between all factor levels. Plots 3.4 *b* and *c* suggest that elevation and placement on the shore do not influence community structure, or are not the dominant factor.

Table 3.1: Total numbers of species, their mean percentage covers and mean species diversity (D) on internal and external surfaces. Where 'n' = the number of times a species was recorded for each exposure factor level and * denotes species <0.1% mean cover.

		Internal				External			
Group	Species	Total no. species	n	Mean % cover	SD (±)	Total no. species	n	Mean % cover	SD (±)
Green Seaweeds		1				1			
	Ulva spp.		34	4.8	8.8		60	47.4	31.4
Red Seaweeds		6				2			
	Mastocarpus stellatus		15	0.4	1.1		15	0.3	0.7
	Chondrus crispus		28	7.9	15.9		4	1.3	5.2
	Hildenbrandia spp.		2	0.1	0.6		0	-	-
	Erythroglossum laciniatum		1	0.1	0.6		0	-	-
	Porphyra spp.		1	0.0*	0.1		0	-	-
	Polysiphonia spp.		2	0.0*	0.2		0	-	-
Brown Seaweeds		3				2			
	Fucus spiralis		6	0.5	3.3		9	0.7	2.4
	Algaozonia		1	0.2	2.2		0	-	-
	Dictyota dichomota		0	-	-		3	0.1	0.6
	Sargassum muticum		5	1.9	10.3		0	-	-
Invertebrates		10				5			
	Patella vulgata		27	0.6	1.2		26	0.9	1.7
	Patella depressa		14	0.3	0.6		23	0.5	0.9
	Patella ulyssiponensis		7	0.1	0.5		12	1.6	5.8
	Actinia equina		19	0.4	0.9		0	-	-
	Actinia fragacea		1	0.0*	0.1		0	-	-
	Eulalia viridis		3	0.0*	0.2		0	-	-
	Carcinus (juv)		1	0.0*	0.1		0	-	-
	Mytilus edulis		35	0.7	1.4		15	0.3	0.8
	Nucella lapillus		4	0.1	0.2		0	-	-
Total cover		20	236	24.7	61.8	10	175	53.5	50.8
Simpson's Index of Diversity (D)			0.90 (±0.002)				0.82 (±0.006)		

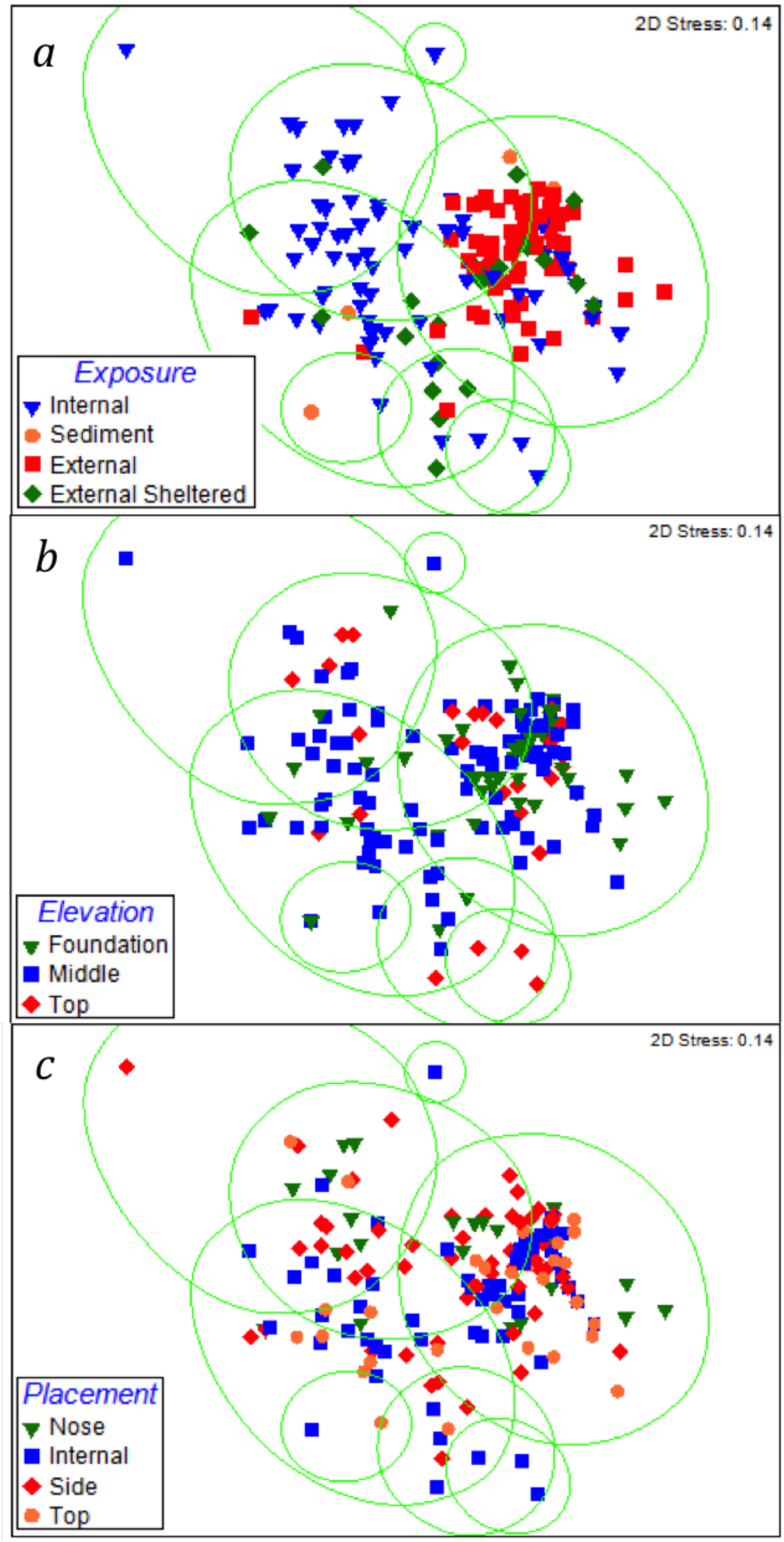

Figure 3.4: MDS plots of data based on rank similarity for (a) exposure (b) elevation on the shore (c) placement within the structure (Fig B). Contours of 25% similarity are shown.

PERMANOVA analysis used to test multivariate species assemblages in response to the three factors recorded, and more specifically the impacts of exposure level on communities. Table 3.2 displays the results of the PERMANOVA analysis for each factor (exposure, elevation on the shore and placement within the structure) and interactions between factors, shown in the 'Factor/ Interaction' column; and the results of the pairwise comparison tests for exposure level (the factor of significant interest). For each factor, interaction and pairwise comparison, a 'Pseudo-F' or 't' value is calculated based on the variation within the data, and the P value (significance denoted by the number of *) explains the significance of the factors, interactions and tests. PERMANOVA highlighted significant differences in the species assemblages due to exposure (Pseudo-F_2 8.80, $P = 0.001$), elevation (Pseudo-F_2 3.77, $P = 0.001$) and placement (Pseudo-F_3 2.44, $P < 0.01$). Significant differences in species assemblages due to elevation and placement reflect different outputs to the MDS plots, however, interpretation of MDS plots is a qualitative technique of visual patterns and a lack of pattern in the MDS plots could be due to the 2D representation not showing potential patterns which would otherwise be displayed if visualised in 3D.

Additionally, PERMANOVA showed significance between the interactions of exposure and placement factors (Pseudo-F_6 2.14, $P = 0.001$), and exposure and elevation (Pseudo-F_4 2.16, $P = <0.01$), but no significant impact on species assemblages due to the interactions of placement and elevation, or all factors combined. The interactions displaying significant difference between species assemblages both include the exposure factor potentially suggesting this is the dominant factor, however it is clear that all factors influence species assemblages individually or combined.

Further analysis of the exposure factor using pair-wise tests showed significant differences between external areas and other exposure levels, particularly internal and external (t = 5.20, $P = 0.001$), and sediment and external (t = 2.78, $P = 0.001$), which are comparisons of the most exposed levels and sheltered levels, and therefore would be expected to display strong differences. Notably, external areas compared to sediment areas would logically provide stark contrast as no species were recorded within the sediment areas. More specific analysis was not carried out on other factors (elevation and placement) as these were not the focus of the study, However, these factors were included within the experimental design in order to gain a comprehensive view and account for other potential impacts on the communities when analysing.

SIMPER analysis (Table 3.3) confirmed that *Ulva* spp. were the characterising organisms causing observed differences between internal and external faces contributing 42% of the dissimilarity observed. *Ulva* spp. were recorded on every external surface and covered the surface faces, whilst other species were found colonising smaller areas.

Table 3.2: PERMANOVA analysis identifying the impacts of the physical factors that affect community colonisation and species richness (*** *P(perm)* = 0.001; ** *P(perm)* <0.01; **P(perm)* <0.05.

Factor/ Interactions	Pairwise test	Pseudo-F	df	t
Exposure		8.80***	2	
	1. Sediment, External			2.78***
	2. Sediment, External sheltered			1.34
	3. Sediment, Internal			1.00
	4. External, External sheltered			1.92*
	5. External, Internal			5.20***
	6. External sheltered, Internal			1.44
Placement		2.44**	3	
Elevation		3.77***	2	
Exp x Place		2.14***	6	
Exp x Elev		2.16**	4	
Place x Elev		1.44	3	
Exp x Place x Elev		1.11	3	

Chondrus crispus (14%) and barnacles (13%) were contributing factors but were found in higher abundance on internal compared to external surfaces. In addition, SIMPER analysis carried out on other factors showed *Ulva* spp. to be the dominant contributor to dissimilarity for all factors and at all levels. Internal levels for placement and Middle level for elevation factors showed smaller contributions from *Ulva* spp. which reflects the results of the exposure factor as these are both more internally orientated levels on the structure compared to more exposed areas such as the Tops and Sides. These could account for the significant results between factor interactions calculated in PERMANOVA.

3.3.2 Maintenance Disturbance

Average disturbance due to the maintenance works was recorded and calculated as a percentage of the surface cover, the number of occurrences and the extent of the damage as a percentage of the total number of samples for each factor level, to consider the extent of engineering disturbance on communities. Table 3.4 presents the results recorded and shows the external level of the exposure factor, side level of the placement factor and middle level of the elevation factor each had the highest incidences of where maintenance damage observed and the highest average percentage cover per unit. External sheltered level from exposure factor and nose level from the placement factor had the highest number of occurrences as a proportion of the total sample. The percentage of the total sample was important to consider as there were different numbers of sampled units

Table 3.3: Results of SIMPER analysis reporting the percentage (%) contribution of species to assemblage dissimilarities for exposure levels on the groyne (internal and external) and similarities for the internal and external levels. Only species with contributions higher than 3% in at least one pair-wise comparison are reported. Numbers in brackets are average dissimilarities (Int x Ext)/ similarities (internal and external) between assemblages.

Species	Int x Ext (76.14)	Internal (24.67)	External (59.44)
Ulva spp.	42.72	29.61	94.43
Chondrus crispus	13.51	20.14	-
Barnacles	13.30	22.06	-
Patella vulgata	5.43	7.4	-
Mytilus edulis	5.01	11.39	-
Mastocarpus stellatus	3.44	-	-
Patella depressa	3.39	-	-
Actinia equina	3.38	-	-

per level due to their location on the groyne structure. The results suggest maintenance disturbance recorded from areas which were general externally facing or more exposed occurred much more frequently and caused a higher amounts of damage to the surfaces. A chi squared (X^2) test for the number of occurrences of maintenance disturbance on the structure shows significant differences between occurrences across each factor (exposure: P = 0.001; placement: P <0.05; elevation: P <0.01).

Table 3.4: Average percentage cover of maintenance disturbance, number of occurrences of damage on internal and external rock armour units, and the frequency of occurrences (%) out of the total number of faces sampled (280).

Factor	Level	Average Disturbance (% cover)	No. of Occurrences	% of Total Sample
Exposure				
	Sediment	0	0	0.0
	Internal	1.36 (±5.98)	8	7.3
	External sheltered	1.96 (±6.21)	5	20.8
	External	2.81 (±8.19)	15	14.2
Placement				
	Nose	2.76 (±6.75)	8	21.1
	Internal	0.84 (±5.44)	4	3.3
	Side	2.95 (±8.39)	13	15.5
	Top	1.11 (±3.98)	3	8.3
Elevation				
	Foundation	0.45 (±1.94)	5	5.6
	Middle	2.50 (±8.36)	16	11.8
	Top	2.09 (±6.21)	7	12.7

3.4 Discussion

3.4.1 Habitat Provision

The removal of a porous coastal defence structure provided a unique opportunity to gain better insight into the total habitat provision capabilities of artificial structures. It is unusual to come across the decommissioning of coastal defence structures and this rare opportunity provided access to areas of artificial structures that have not previously been investigated or actively considered as a potential suitable habitat for coastal assemblages. By carrying out biological and physical sampling during removal of a porous defence structure, we were able to gain important insights into the coastal species found on artificial structures.

We found significant differences between the biological communities present in internal and external environments (Table 3.2 and Figure 3.4). Internal surfaces supported twice as many species of both invertebrates and algae as the external environment, particularly mobile species (Table 3.1), therefore demonstrating higher species richness in internal habitats on porous defence structures. Despite showing differences in the values recorded for species diversity total percentage cover between internal and external environments, these were found to be insignificant. However, further analysis (Table 3.3) indicated differences between species within the communities with *Ulva* spp. being the dominant species externally, and overall across all factors. *Ulva* spp. are green ephemeral opportunistic early successional species and require light to survive. They covered much of the rock unit surfaces in the external environment. This was most likely because of the unfavourable conditions for the majority of invertebrate species, and lack of grazing pressure (Hawkins, 1981; Jenkins et al., 2005; Coleman et al., 2006). There was only one species that was found to colonise the external exposed surface and not the internal surface which was *Dictyota dichotoma* (brown fan weed). Invertebrate species were mainly found to colonise the internal areas of the structure, which was likely sought after as a refuge against wave exposure and to allow for foraging, whilst avoiding scour, wave exposure, desiccation and potential predation (Silva et al., 2008).

The results between the PERMANOVA analysis and MDS plots for the factors Elevation and Placement (Table 3.2 and Figure 3.4 b and c), showed conflicting results. MDS plots indicated a lack of differences between the species assemblages due to the high dispersals in the plots and little observational pattern. However, PERMANOVA showed significant differences in the species assemblages because of elevation and placement factors. Inconsistencies between the results of the two tests are likely due to the qualitative nature of MDS plots and the visualisation of them on a 2D plane, which could mask potential patterns which would otherwise be displayed in 3D. PERMANOVA is a robust test for hypotheses testing, whereas MDS plots provide observational data, therefore it is accepted that there are significant differences between species assemblages due to all three factors. The results also showed interactive effects on species assemblages due to exposure and elevation, and exposure and placement. This suggests that exposure is perhaps the most dominant factor effecting community compositions, whilst levels within other factors play a part. It is also likely that certain levels within each factor are likely to be recorded together, for example, more external rock units will inevitably only be located at certain locations related to elevation or placement within the structure, such as the nose or sides of the groyne, compared to the internal exposure levels which would not be categorised under those locations.

The results of our study support part of our first hypotheses (1) that internal habitats on the porous defence structure will support greater species abundance than external habitats; and (2) internal habitats will support a higher number of invertebrate species than external habitats. Coastal defence structures are constructed in high dynamic environments where there is increased pressure on invertebrate and plant species. *Ulva* spp. are known to colonise marine intertidal habitats (Bunker et al., 2010; Maggs et al., 2007) and dominate exposed surfaces leaving very little surface for other species to attach and colonise, therefore creating inter- and intraspecific competition for space and reducing biodiversity in the areas where *Ulva* spp. colonise. *Ulva*, a green algae species, has high light requirements for photosynthesis, therefore colonising external, and non-shaded areas (Bunker et al., 2010; Maggs et al., 2007). *Chondrus crispus* and *Mastocarpus stellatus* successfully colonise the internal areas more than the external environments. Although these species often colonise exposed natural rocky shores, they can tolerate reduced light levels (sciaphilic) (Bunker et al., 2010) and are therefore able to colonise shaded, internal areas where there is more protection from waves. They are also later successional species and may be excluded by persistent ephemerals (Sousa, 1979). Invertebrate species were found to colonise primarily the internal areas, with very few (low cover) exceptions. Species distributions on rocky shores are set by the interplay of vertical (tidal elevation) and horizontal (wave action) stress gradients, coupled with biological interactions (Raffaelli and Hawkins, 1996). Refuges are provided by microhabitats created by crevices, cracks and rock pools, which are common features of natural rocky shores (Johnson et al., 2003).

Until now, artificial structures have been perceived as poor surrogates for natural shores because they lack habitat complexity and heterogeneity (Chapman and Blockley, 2009; Moschella et al., 2005; Firth et al., 2013b,c). Our study shows that porous defence structures can provide valuable habitats for species to colonise formed between rock unit interfaces providing potential refuge from desiccation stress (Hawkins and Hartnoll, 1983) and disturbance through scouring by cobbles, gravel and sand (Moschella et al., 2005; Bulleri and Chapman, 2010). Not only can porous defence structures effectively dissipate wave energy onto the coastline (Silva et al., 2000; Garcia et al., 2004; Jung et al., 2012), serving an important engineering function, they may also provide habitat complexity and protection within their interstices that encourage higher biodiversity than other types of coastal defence designs. They also enable water flow within the structure, providing access to food and submersion for periods, which is essential for many intertidal species. Thus porous defence structures are a desirable and multifunctional type of coastal defence structure.

3.4.2 Maintenance Disturbance

There has been little research investigating the effects of maintenance disturbance on coastal assemblages (but see (Airoldi and Bulleri, 2011). Our study demonstrates the levels of disturbance that occur during coastal maintenance, particularly to internal and external environments. Anthropogenic disturbance can create openings for opportunistic and invasive species to settle (Dayton, 1971; Sousa, 1979; Hutchinson and Williams, 2003). One key finding is the difference in disturbance levels between internal and external environments. There was nearly double the amount of maintenance disturbance on the external rock unit faces compared to internal ones. Typical coastal maintenance will often involve the replacement of a number of rock units that may become dislodged or moved during intense weather conditions. This will mainly be on external rock units that are more exposed to the extreme conditions and susceptible to movement. This not only creates potential weakness to the structure and its function, but also disrupts species occupying the units moved and any associated connecting units. The internal environment, however, is protected from this disturbance. This emphasises the importance of internal environments as suitable habitats to support higher levels of biodiversity on coastal shores. Future work should be carried out to further investigate the effects of disturbance.

Overall, coastal managers should consider the design of structures not only for engineering function, but environmental benefits. This study shows that rip rap designs can provide more environmental benefits over solid based structures due to their porosity and internal cavities provided, therefore creating multifunctional defence structures. Further studies should be carried out in order to provide more evidence supporting the role of porous defence structures as suitable surrogate habitats for natural shores, in addition to more extensive work on maintenance disturbance. With further evidence supporting porous CDS designs and demonstrating the environmental benefits, reduction in maintenance work and associated costs by using these design, policy makers should consider advising the use of porous structures where possible within coastal defence schemes.

3.5 Conclusions

Until now, the internal environment of CDS has not been actively considered or explored by ecologists for its potential to provide habitat and enhance biodiversity. My study highlights the importance of these hidden environments for coastal species, suggesting that porous CDS provide improved habitat heterogeneity and refuges via internal compartments. These features are not present in solid structure designs with no internal compartments. External environments on coastal defence structures are exposed to intense environmental pressures made worse by anthropogenic disturbance from any maintenance work. Therefore they only support a small number of hardy species. Focus must be turned to the internal environment, which can support a higher diversity of species. Porous structures, a common coastal engineering design, are not only effective in engineering; they are also effective for biodiversity. Porous CDS should be considered more widely in future coastal engineering schemes, to encourage settlement of coastal species and sustain coastal communities, particularly on the growing number of artificial structures. Finally, further investigations into the impacts of maintenance activity on coastal assemblages should be considered in order to inform coastal engineers and provide evidence-based decisions for effective management regimes for coastal defence.

Chapter 4

Additive Manufacturing for Coastal Engineering

Abstract

The demand for more environmentally sensitive coastal defence structures has led to innovative designs in ecological enhancements. Current methods show increases in species diversity on artificial surfaces with structural enhancement features because of increased habitat complexity. However, these techniques use traditional engineering materials with many design constraints. 3D printing technology provides opportunities for more sustainable and advanced coastal defence structure designs through its flexible and novel manufacturing process. This study looked at the biological colonisation of forty-four *D-Shape* 3D printed tile units of five varying designs, deployed at Highcliffe, UK. The material demonstrated some biological settlement in the marine intertidal zone on two of the tile units, mostly *Ulva* spp. However, 3D printing shows potential constraints in the design specification when manufacturing for harsher marine intertidal environments. Fracturing, erosion and loss of forty-two tiles was recorded. Further large scale studies could provide more substantial and positive evidence for the use of 3D printing in maritime engineering, with considerations to scale of design features.

4.1 Introduction

A drive in construction efficiencies has fuelled the rapid progression of automated technologies, including *Additive Manufacturing* (AM) (Lim et al., 2012). AM is defined by the American Society for Testing and Materials (ASTM) as "...the process of joining materials to make objects, usually layer by layer, from 3D CAD data" (ASTM, 2012).

3D printing more specifically, is an AM technology used to print 3D solid objects from digital models (Bogue, 2013). Since 2013, the application of AM has extended from predominant use as a rapid prototyping tool, to a broader range of manufacturing capabilities supplying to a variety of disciplines and producing a diversity of materials (Evans and Ian Campbell, 2003; Sambu et al., 2004; Lim et al., 2012; Bogue, 2013).

Previous research on the ecological enhancements to coastal defence structures (CDS) shows increases in species diversity on artificial surfaces with structural enhancement features, compared to those without (Chapman and Blockley, 2009; Martins et al., 2010; Browne and Chapman, 2011; Firth et al., 2012; Perkol-Finkel and Sella, 2015; Ido and Shimrit, 2015; Evans et al., 2016). However, these enhancements use traditional engineering materials, such as limestone, granite and concrete, (Moschella et al., 2005; Burt et al., 2009b; Coombes et al., 2011; Firth et al., 2012, 2014; Evans et al., 2016) with high acidity levels, intense carbon footprints and low topographic complexity (Allen, 1998; CIRIA, 2010; Coombes et al., 2015; Ido and Shimrit, 2015). This, combined with the relatively expensive, basic, labour intensive and (often) unreliable methods of implementing features, means there is a gap in the market for an efficient, cost effective, and intricate method of building complex features. The development of new and efficient processes, and sustainable materials would benefit many coastal engineering schemes.

4.1.1 *D-Shape* Technology

Created in 2007, Monolite UK Ltd owns the world's first and largest construction scale 3D printing technology, *D-Shape* (Figure 4.1). The *D-Shape* technology differs substantially from other large-scale additive manufacturing methods in terms of both materials produced and technique (Jakupovic, 2013). The printer uses crushed dolomitic limestone aggregate (or other aggregates) together with a patented magnesium chloride binder to create a material known as *Sorel Cement Chemistry* (Jakupovic, 2013; Dini, 2014) (see Appendix D for technical details). Operated through a Computer Aided Design and Computer Aided Manufacturing (CAD-CAM) system, the *D-Shape* printer binder is deposited in specific areas identified through the digital design to create a solid part. This process continues in 5 mm layers, with solidification between each one, until the full product is achieved (see Figures 4.2 for time series of digital to deployed product). The final material comprises approximately 70-75% aggregate, 10-25% binder and the rest is void spaces. Tested through a series of development experiments, the binder chemical properties make the material environmentally neutral (no leaching or impact on pH) and suitable for use in marine environments (Figure 4.2 & 4.3) (Dini, 2014). Previous studies show the mechanical characteristics and strength properties of the *D-Shape* printed materials are adequate for use in construction (uniaxial stength = 54.5-58.7

N/mm^2; flexural strength = 13.3-14.7 N/mm^2; static modulus of elasticity 24853-25452 N/mm^2) (Jakupovic, 2013). Although this technology has proven suitable as a construction technology, there are a number of considerations that should be acknowledged when designing structures. Table 4.1 outlines a number of printing specifications for designs.

Table 4.1: Specifications and considerations to design items to be 3D printed with *D-Shape*.

Design Consideration	Specification	Note
Binder ink capillary accuracy	+ 10 mm	Add 10 mm to all designs due to ink spreading
Printing a wall	Thickness >20 mm	Avoid brittle walls
Printing a cavity	Circular cavities >80 mm diameter	Reduction in cavity size by 20 mm due to capillarity spreading
Printing a complex shape	Thickness >30 mm	Can compromise strength of the material
	Cylindrical shapes >30 mm	Increase in protrusion size by 20 mm due to capillarity spreading
Intricate internal features	Large enough external entrance	Remove the loose substrate from the inside

The practicalities and variety of sand/ cement based materials (Lim et al., 2012) makes 3D printing an ideal technology for incorporating innovative design and construction methods, with sustainability and ecological enhancement features in coastal engineering schemes. Combining this process with other new and emerging technologies such as 3D laser scanning and complexity development software (Loke et al., 2014, 2015), means more flexible and advanced designs are possible. To date, there have been a limited number of studies testing the biological capabilities of the *D-Shape* technology as a marine environmentally sensitive material, including two artificial reef projects deployed in Bahrain, Sydney harbour and Larvotto Underwater Reserve, Monaco (Figures 4.2) (Dini, 2014; Boskalis, 2016). However, no work has been done investigating the use of this technology in the maritime engineering industry, particularly CDS. This would test not only the ecological benefits but also the construction and strength capabilities in a harsher coastal environment.

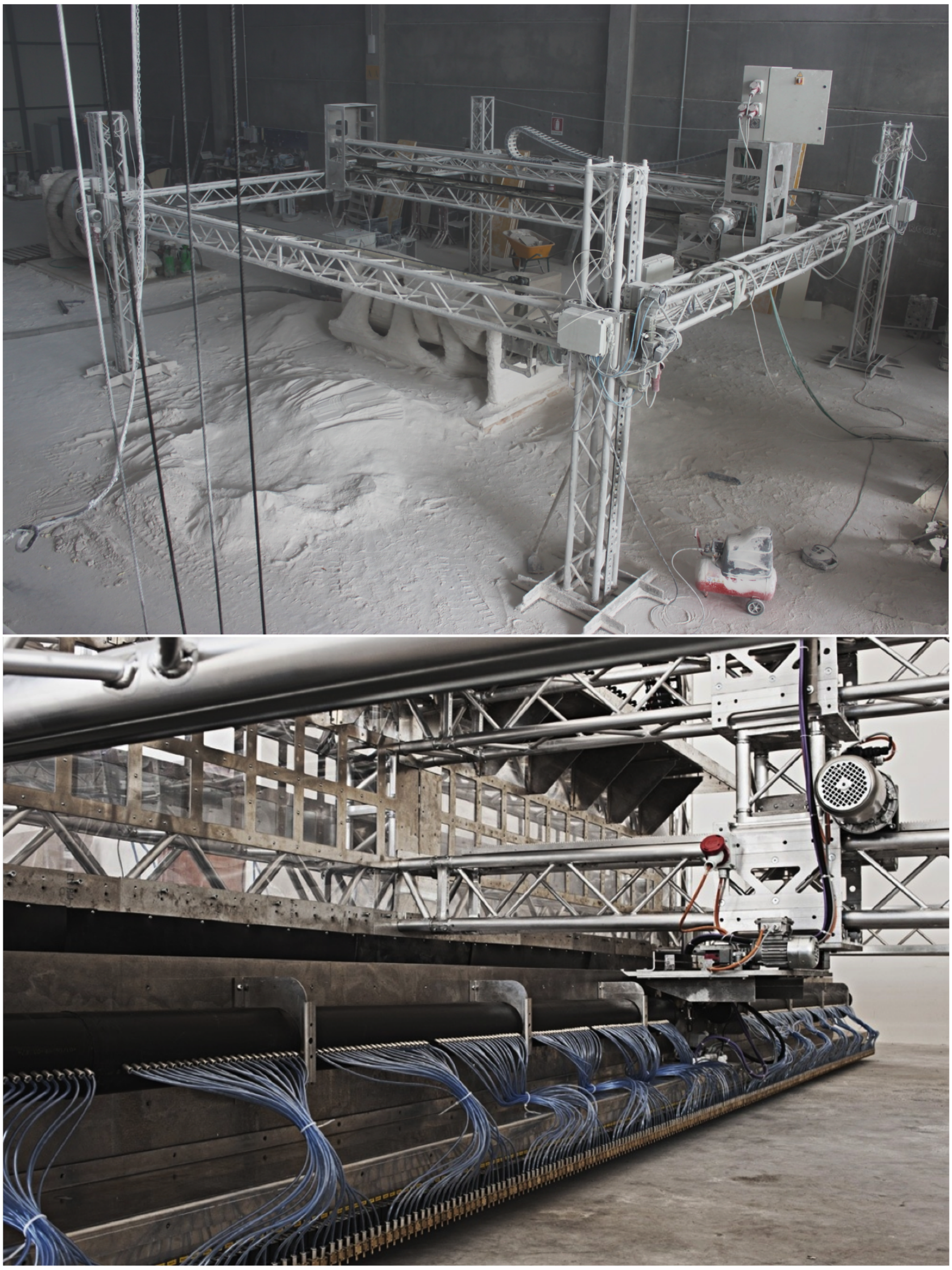

Figure 4.1: D-Shape® 3D printer and ink jets in Italy. Image courtesy of Monolite UK Ltd. Top image shows printer scaffolding (6 x 6 m area) and runners, allowing printer to operate in a 3D parameter (horizontally and vertically) along the scaffolding. Bottom image presents the rows of 300 ink jet nozzles.

Figure 4.2: Example of 3D printed reef unit constructed by *D-Shape* from computer designing, printing and then field deployment in Bahrain. Images courtesy of Monolite UK Ltd.

Figure 4.3: 3D printed test ring deployed to determine the materials resistance to salt-water. Image courtesy of Monolite UK Ltd.

4.1.2 Aims and Objectives

In this chapter we investigate if:

1. there will be a significant increase in the abundance of biological species over time on the 3D printed material;
2. will *D-Shape* 3D printed tile units withstand exposure in a marine intertidal environment.

4.2 Methodology

All test materials were supplied by Monolite UK, facilitated by Tekint® and printed using the *D-Shape* 3D printer technology based in Pisa, Italy. All tile unit designs are the intellectual property of the University of Southampton as part of this thesis and were supplied to Monolite UK and Tekint® where they were digitally converted to comply with the *Monolite* programme. Forty-four 3D printed tile units were produced with five different designs of increasing complexity (Figure 4.4 & Table 4.2). Tile designs were created to consider niche habitat spaces available on natural rocky shores in which intertidal species seek refuge, e.g. rock pools simulated by Figure 4.4 E, and crevices

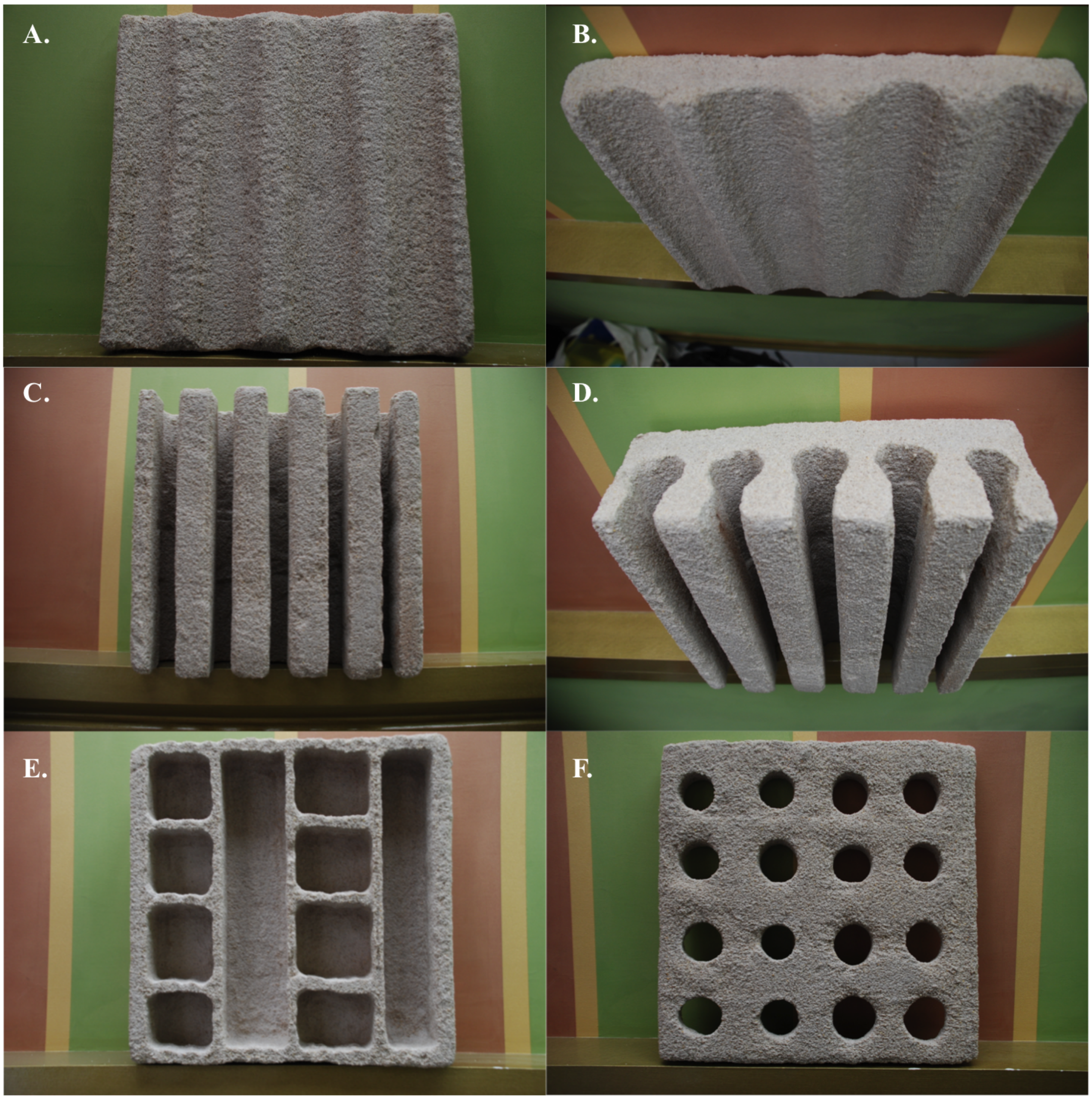

Figure 4.4: 3D printed tile units with five different designs of varying complexity. A & B = *Rough* design, C & D = *Wiggles*, E = *Squares*, F = *Circles*. Image supplied by Monolite UK Ltd.

simulated using the 'Wiggle' design in Figure 4.4 C & D. In addition, less drastic features were designed to simply increase surface roughness and texture in order to see if there is any preferences in tile design to biological species, and engineering feasibility (i.e. are more complex designs realistically able to be added to engineering structures or is there a limit to the design complexity available).

Table 4.2: Dimensions (mm) and number of replicates used of each 3D printed tile unit design. Designs increasing in surface texture and design complexity from 'Control' being least complex to 'Wiggles' being the most complex and drastic design. Refer to 4.4 for further details.

Design	Replicates	Length	Width	Height	Design Description
Control	10	300	300	30	Plain, no design
Rough	10	300	300	30	Mildly uneven surface texture
Holes	8	300	300	30	Flat tile with circular holes perforating through the unit
Squares	8	300	300	50	Raised walls and partitions providing square and rectangular water retaining compartments
Wiggles	8	300	300	50	Drastically uneven surface with deep cavities

4.2.1 Study Site

The study took place at Highcliffe in Christchurch Bay on the south coast of England, UK (see Chapter 3 for full site description). Forty-four tiles were deployed on two far western rock groynes along the low to high water level, and around the east, west and nose of the groynes (Figure 4.5). Tiles were attached using four marine steel bolts (5.5 x 160 mm) and washers (6 mm diameter) per tile drilled into the faces of the rock armour units (one per each corner). A waterproof epoxy resin (EPOMAX®) was used to seal and secure the bolt-holes (Figure 4.6). Placements of tile designs was selected using a random coordinate generator with the following conditions (see Appendix E for details):

1. No more than one tile per face;
2. Every tile design had to have a minimum of one tile at:
    - each level on the structure (low, medium or high water level);
    - each side of the structure (east, west and nose) per groyne;
    - vertical and horizontal orientation

4.2.2 Data Sampling

Tiles were deployed in April 2015, and sampled every two weeks for the first two months, and then monthly thereafter for a further five months (totalling seven months of recording). Each tile unit had the following identification information recorded: the groyne

Figure 4.5: Image of the groyne system constructed at Highcliffe within Christchurch bay. Image shows the eleven rock groynes and a bastion of varying long and short lengths, and highlights the two study groynes where the tile units were attached. Image adapted from Imagery ©2016 Google, TerraMetrics, Map data ©2016 Google.

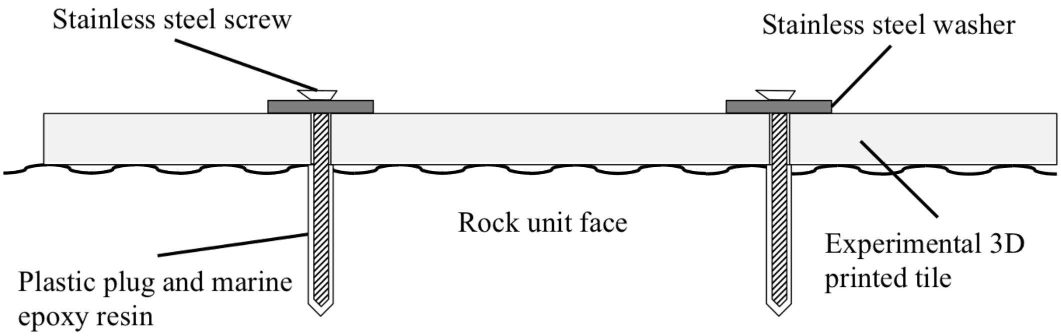

Figure 4.6: Side view of tile fixing to rock armour face using A4 grade stainless steel screws and washers, and EPOMAX marine epoxy resin.

number, sample number, design, elevation on the structure and orientation of the tile unit (see Appendix 3). Photographs and physical details of the location of each tile unit were recorded along with biological and physical data. Biological sampling was conducted for each tile unit by identifying organisms present to species level where possible with counts for mobile fauna and percentage cover. Where it was not possible to identify to species level, identification was made to the closest taxonomic level. Physical data included notable changes in weather conditions including strong storms that had occurred; any displacement to the groyne structures; and the number of tile units lost/ destroyed/ damaged.

4.2.3 Statistical Analysis

To test our first hypotheses, total species abundance based on percentage cover was pooled for each tile unit (n = 44) and plotted over the seven months time period. Further biological analyses was carried out using PRIMER-E ver. 6 and PERMANOVA statistical software (Clarke, 1993; Anderson et al., 2008; Clarke et al., 2014) to determine the difference in species abundance over time on each tile unit. This experimental design consisted of four fixed factors: groyne number (two levels: G1 or G2), tile design (five levels: control, rough, holes, squares and wiggles), elevation on the structure (three levels: low, middle or top), and orientation of the tile (three levels: horizontal, vertical or slanted). Multi-dimensional scaling (MDS) plots based on a Bray-Curtis similarity matrix (Bray and Curtis, 1957) of square-root transformed data were created for each factor (groyne number, tile design, elevation on the structure, and orientation of the tile) to visualise data patterns using rank similarities and hierarchical clustering in the multivariate output (Clarke, 1993; Clarke et al., 2014). Permutational Multivariate Analysis of Variance (PERMANOVA) was then used to test multivariate species abundance based on 9999 unrestricted random permutations of residuals (Anderson et al., 2008). A four-way factorial design was used with the four factors specified previously. To address our second hypothesis, the total number of tile units remaining was pooled (n = 44) and plotted over the seven months time period. A Chi-squared (X^2) test was then used to determine if there was a significant loss of tile units, based on the assumption that there should be no loss of units.

4.3 Results

Forty-four (44) tile units of five different designs were deployed across two rubble mound groynes at Highcliffe on the South-coast, UK. Figure 4.7 shows examples of the deployment and fixing of each tile design using the stainless steel bolts. Over the seven month period there was a significant loss of the 44 tiles with only two remaining and large amounts of damage was observed before dislodgement. A chi squared (X^2) test shows a significant loss in tiles overall ($P < 0.001$). Figure 4.8 shows the rate of loss of tile units over the seven month recording period, with a loss of 41% of tiles within the first month, followed by the second largest losses in the second and third months. The remaining two tiles both had the rough design, located at the mid shore level and orientated in a horizontal placement. However, they were situated on the east and west of the same groyne, therefore receiving different types of wave exposure (east facing sides generally receive greater environmental exposure and disturbance). Physical damage to every tile was observed, notably significant erosion on the sea-facing edges which showed no signs of colonisation (Figure 4.9), and cracks through the tile units, particularly on those with

Figure 4.7: 3D printed tile units deployed and sampled for species growth at Highcliffe, UK.

a smaller height and no walls (e.g. control, rough and circles - Figure 4.10). In addition, most tile units showed clear signs of erosion around the screw-holes (Figure 4.11) and where tiles had been lost the screws were still present in the groynes (Figure 4.12).

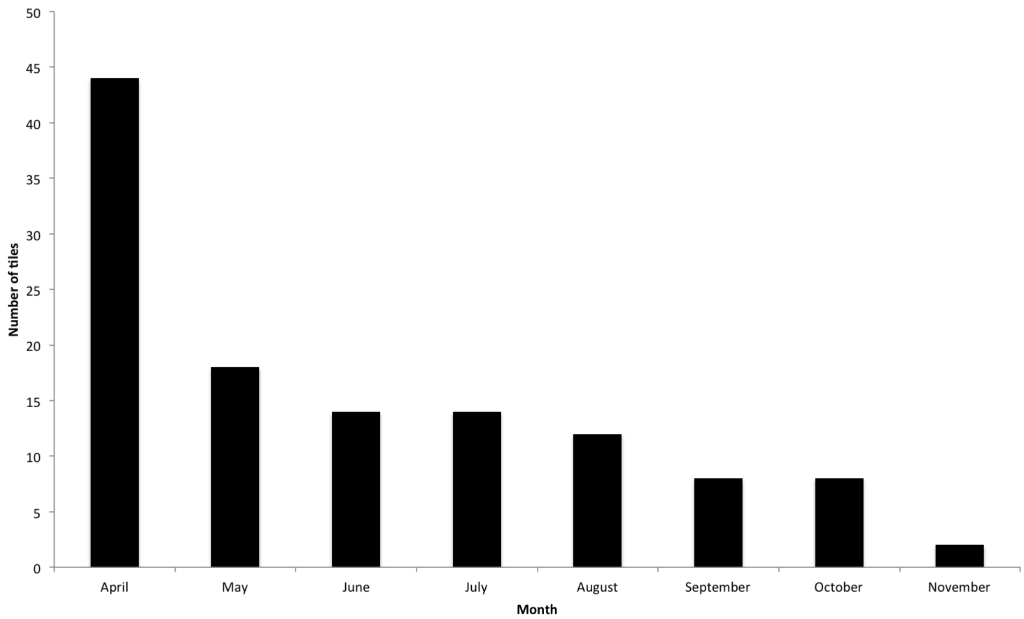

Figure 4.8: Number of 3D printed tile units remaining over the seven month period.

Figure 4.9: Erosion on the seaward faces of the tiles due to wave attack. Images taken at 1 month after deployment.

Figure 4.10: Cracking and breaking of the tiles due to wave energy. Image taken 1 month after deployment.

Figure 4.11: Erosion around the screw fixings on the tile units.

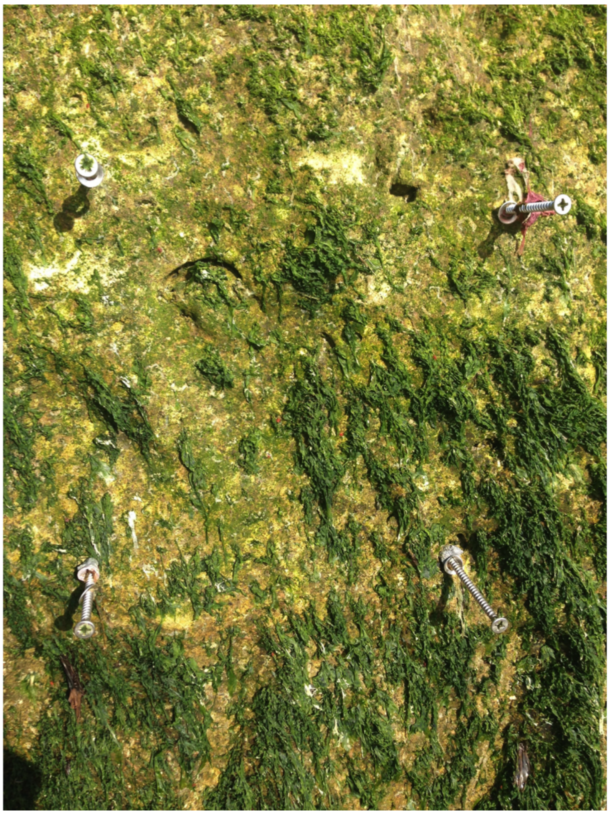

Figure 4.12: Screws remaining on the rock units after tiles have been dislodged.

At time-zero, there was zero species colonisation on each of the 44 tile units, in contrast to the highly covered rubble mound units they were attached to, dominated by *Ulva* spp. Colonisation of the tiles as an accumulation of the total percentage cover of all species recorded, showed overall a rapid increase on the western tile, and a slower colonisation of the eastern tile. Both tiles achieved ∼100% colonisation over the seven months (Figure 4.13 & 4.14), however, this was limited mainly to *Ulva* spp., similar to the limestone groyne rock units. Species of foliose red algae were observed to colonise a number of the tiles (<5%), notably *Osmundea Pinnatifida* and *Mastocarpus stellatus* on the square design, and *Chondrus crispus* found on one rough tile. In addition, one *Patella vulgata* was observed on a rough tile unit, and signs (surface scar) of a further limpet were noted but could not be formally included in the recordings. Due to the significant loss of tile units, it was not possible to carry out formal statistical analysis on the biological data as initially intended.

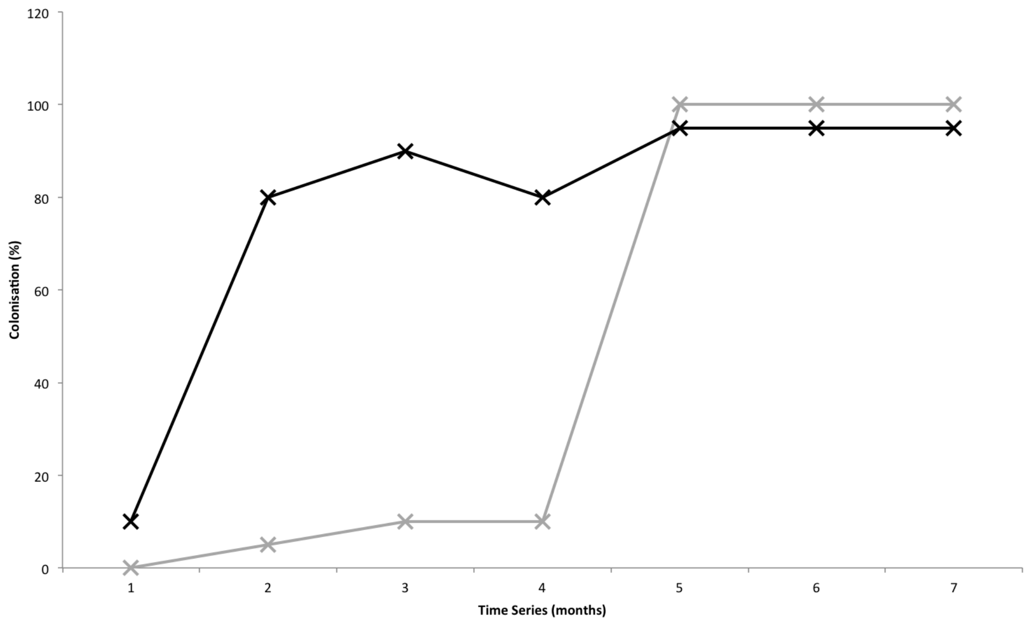

Figure 4.13: Percentage cover of species settlement on the 3D printed tile units over the seven month period. Black line represents tile located on the west side of the groyne, and grey is the tile on the east. n = 1 for each tile.

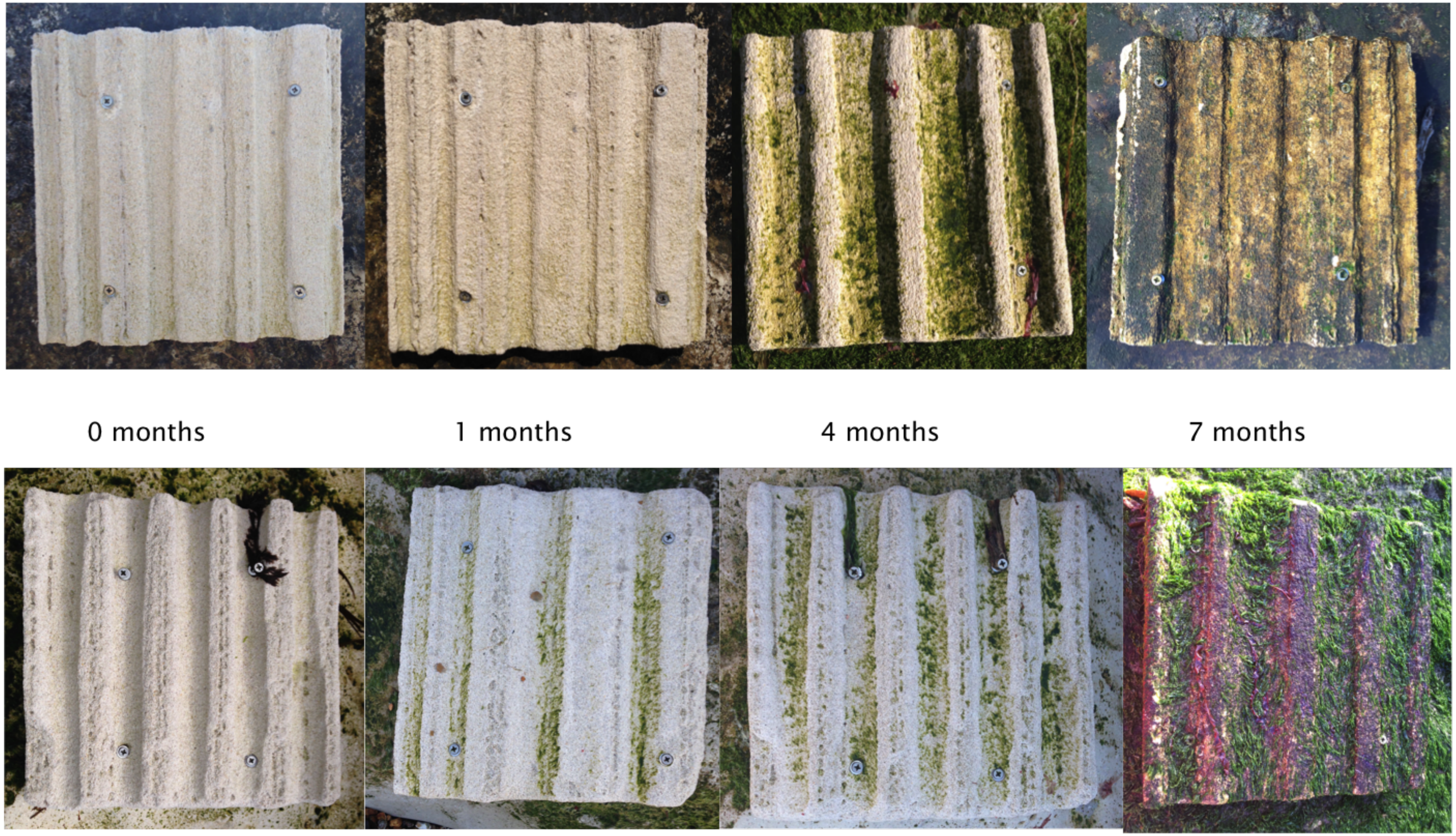

Figure 4.14: Biological colonisation recorded on the 3D printed tile units over the seven month period.

4.4 Discussion

The deployment of 3D printed tile units showed rapid colonisation over a short recording period of only seven months. Although only two tiles out of the total of forty-four that were deployed were remaining, they both displayed nearly 100% colonisation. Both tiles showed mostly *Ulva* spp. colonisation, but also recorded one limpet (*Patella vulgata*). *Osmundea pinnatifida* and *Mastocarpus stellatus* were recorded on square tile, and *Chondrus crispus* on a rough tile unit, all of which were lost. Overall, the biological colonisation of species observed in this study was clearly lower than those observed in other studies that have implemented design modifications (Chapman and Blockley, 2009; Martins et al., 2010; Browne and Chapman, 2011; Firth et al., 2012; Perkol-Finkel and Sella, 2015; Ido and Shimrit, 2015; Evans et al., 2016), however, this could be due to the the short time-frame of the experiment due to the loss of tile units and therefore it is not possible to fully compare the use of 3D printing compared to other modification techniques. There was no pattern in the loss of tile units due to their design, although, tile units placed lower on the shore were lost more rapidly than higher locations, which would be expected due to the higher levels of inundation and dynamic environment experienced.

There was notable damage to all tile units due to erosion, particularly on seaward facing areas, and fractures. Furthermore, where tile units were lost, there was the remains of the screws, and those which endured a longer period showed clear indications of erosion around the screw holes. This suggests that while the *D-Shape* material has suitable strength characteristics for use in construction (Lim et al., 2012; Jakupovic, 2013), there are still a number of limitations to the technology, in particular its use in maritime engineering. The most likely factor is the scale and refinement of the specific design criteria. Monolite UK Ltd recommended that designs have a minimum of 20 mm thickness for wall designs, which may be suitable for land based structures, however, it is demonstrated that it is not sufficient in the harsh marine intertidal environment. Further refinement or designs on a larger scale could provide more effective results, given less weakened areas experienced due to the fixing method. Future studies should also consider using alternative fixing methods, which were considered for this study, and testing samples in locations with less extreme environmental conditions and near beaches with less coarse gravel beaches.

Although the results of this study are unable to give conclusive answers to the suitability of the *D-Shape* printed material for use in maritime engineering, the technology still demonstrates potential for designing more intricate details which cannot be achieved through casting or drilling. The study also shows that the material does have good early successional colonisation, however species diversity is not known.

In comparison to tradition materials used for coastal engineering such as limestone, granite and cost concrete, 3D printing provides number of benefits. Concrete is a carbon intensive process and produces acidity levels approximately 7-10% higher than natural rock. In addition, casting is an expensive process and can sometimes be unreliable when incorporating design features as noted by the BIOBLOCK (Firth et al., 2012) which failed after one attempt of casting. The use of quarried rocks such as limestone and granite are often regarded less environmentally friendly due to the quarrying and transportation process, particularly in the UK which granite is a popular material but if often imported from Norway. *D-Shape* technology on the otherhand, produces a much lower acidity, can reduce carbon and environmental footprints but using locally sourced materials and carrying out printing works on site through transport of the printer, thus reducing transportation costs. However, the related costs for 3D printing are generally viewed as more expensive. Whilst traditional material costs can vary from approximately 100-500 m^3 (Keating et al., 2012), *D-Shape* printed material costs are extensively higher at >10,000 m^3 (based on price paid for experimental sample units). This demonstrates the immaturity in the technology currently, however, increased use and application of the technology will likely reduce the costs in future, and overall savings on transportation and labour costs may make it feasible.

4.5 Conclusions

In conclusion, there is still a huge demand for more efficient and environmentally sensitive maritime engineering designs, primarily through their multifunctional ability as protective structures and surrogate habitats. The use of 3D printed technology, principally *D-Shape*, demonstrates a good opportunity to implement more complex and habitat enhancing features into artificial structures. *D-Shape* printed tile units deployed at Highcliffe, UK showed potential for biological settlement, however, there was a severe loss of tile replicates due to the material characteristics and combined storm events. Despite the limitations of this technology, there is still a large gap in the maritime market for a flexible and advanced design methods. Certain design limitations need to be addressed before 3D printing can be a recommended technology in the maritime industry. More notably, the design parameters must be investigated to determine more suitable specifications for us in harsher marine intertidal environmental factors.

Chapter 5

Ecological Engineering for Coastal Protection

Abstract

There is increasing demand for more sustainable and environmentally friendly coastal protection. Wave forces on coastlines play an important role in structuring intertidal coastal assemblages. The use of ecosystem engineering species has been studied in several numerical, laboratory and some field experiments. However, research to date has been limited to a small number of species and has not considered the variation in species properties and their effects on wave energy. In addition, while hard structural enhancement features are designed to promote biodiversity on artificial structures, there has been little consideration of their engineering role in reducing wave energy. This study carried out physical wave studies, investigating the reduction of wave velocities due to the presence of eight different ecosystem engineering mimics, and five different structural surface designs. The results showed significant differences between all of the mimic species, with all of them positively reducing wave velocities. There was also a significant reduction in wave velocities due to increased surface complexity. This study showed that increased rugosity through the presence of species or added structural complexity, can successfully reduce wave velocities. Furthermore it demonstrates the multifunctional role of ecosystem engineers and structural enhancement units by increasing habitat availability and reducing wave velocities.

5.1 Introduction

The increasing demand for more hard and immediate coastal protection due to sea level rise and increased stormier conditions (Marshall, 2001; IPCC, 2007a; Nicholls et al., 2014, 2015) is resulting in continuous maintenance (Airoldi and Bulleri, 2011) due to potential failures (Anderson et al., 2011). Coastlines globally are fringed with a variety of different ecosystems, many of which are biologically rich habitats that provide a wide range of services to human society (UNEP-WCMC, 2011), including natural physical protection (Kamphuis, 2010). They are effective at reducing flooding, erosion (Temmerman et al., 2013; Carus et al., 2016) and the effects due to storm surges (Gedan et al., 2010) through their ability to reduce current velocities and attenuate wave heights. Increasing anthropogenic disturbance through urbanisation, tourism and other human activities are degrading ecosystems globally, resulting in a loss of natural protection (Airoldi and Bulleri, 2011; Salomidi et al., 2013; Perkins et al., 2015) and the associated coastal habitats. Many recent studies have focused on enhancing hard CDS (Moschella et al., 2005; Burt et al., 2009b; Coombes et al., 2011; Firth et al., 2012, 2014; Evans et al., 2016) (see Appendix A for overview). They demonstrate strong evidence for the need to 1) enhance habitat features for coastal species to preserve biodiversity; and 2) protect our coastlines by using more environmentally sensitive designs/ methods. However, these demands could be met through the use of engineering methods which also consider socio-economic benefits, improved aesthetics, natural resources and increased habitat heterogeneity. This chapter looks at *ecology for engineering*, and the role of coastal intertidal and subtidal species as a natural coastal protection method. It also considers the role of hard habitat enhancement features (Appendix A) in engineering through added surface complexity, and thus potentially increasing hydraulic resistance.

The active use of ecosystem engineering species in engineering provides an alternative natural protection and could work alongside hard CDS as buffers to reduce hydraulic storm surges and propagating waves (Dalrymple et al., 1984; Anderson et al., 2011), therefore reducing the size of hard engineering structures (van Leeuwen et al., 2010; Tromp and Van Wesenbeeck, 2011; Borsje et al., 2011). A number of numerical and physical models have looked at the impacts of intertidal and subtidal species on wave energy dissipation (Möller et al., 2011, 2014; Carus et al., 2016), sediment accretion (Meyer et al., 1997; Meadows et al., 1998; Borsje et al., 2011; Gaurier et al., 2011; Mandlier and Kench, 2012) and habitat complexity (Dayton, 1971; Leigh et al., 1987; O'Donnell, 2008) (see Appendix B and Frostick et al., 2014, Table 4.1 for overview). Few studies have been able to effectively implement and quantitatively monitor the use of ecosystem engineers in engineering schemes (Hartig et al., 2011), resulting in a lack of practical evidence and hesitancy from industry practitioners and policy makers to use such techniques over hard CDS. However, projects such as the "Living Shorelines" (Erdle et al., 2006; Swann, 2008; Scyphers et al., 2011; Manis et al., 2015) and "Building with

Nature" (Slobbe et al., 2012; van den Hoek et al., 2012; de Vriend, 2014; de Vriend et al., 2014a,b) have demonstrated ecosystem engineering approaches to design effective flood and coastal erosion management systems for more environmentally sensitive protection. Mendez and Losada (2004) and Anderson et al. (2011) noted that the success of coastal *buffering*, and thus improved protection, through ecosystem engineers is dependent on the characteristics of the species used, such as geometry, buoyancy, density, stiffness, and spatial coverage. A number of studies have used mimic species to replicate and test the impacts of different generalised forms, morphology and textures of species on wave energy dissipation, sediment accretion and habitat complexity, without having to carry out rigorous ethics approval and care of live plants and animals (Kobayashi et al., 1993; Fernando et al., 2008; Augustin et al., 2009; Paul et al., 2012; Hashim and Catherine, 2013; Koftis et al., 2013; Anderson and Smith, 2014). Mimic species allow simplification of real world scenarios, and clear understanding and calculations of the experimental properties, particularly if carrying out scaled physical models (Frostick et al., 2014). Understanding the interactions of different species characteristics and their resistance to different environmental conditions is essential to advise designs of engineering scheme utilising ecosystem engineers. Most research has focused on specific species, or comparisons between taxonomic orders (Fonseca and Cahalan, 1992; Meyer et al., 1997; Möller et al., 2011; Mandlier and Kench, 2012; Möller et al., 2014; Carus et al., 2016), and very few have considered the benefits supplied by different species (Borsje et al., 2011; Manis et al., 2015).

5.1.1 Aims and Objectives

This chapter explores the hydraulic potential of different simulated subtidal and intertidal species (note: not all species included in this study are considered ecosystem engineers but are common species found within coastal regions and therefore deemed of interest to the context of this thesis). Physical hydraulic models were carried out to determine the impacts of eight different mimic species with varying physical properties on wave energy reduction. In addition, experiments were carried out different enhancement designs using tiles over varying surface texture. There has previously been little consideration of the physical presence of hard coastal defence enhancement features on wave properties. Their design to promote habitat heterogeneity, and thus coastal biodiversity, could create hydraulic reducing properties through added complexity. In this chapter I aimed to:

- Compare wave velocity and height with and without the presence of eight different intertidal and subtidal species mimics to determine which specific characteristics are successful at reducing hydraulic loading.

- Compare wave velocity and height with and without the presence of five different structural enhancement units to determine whether surface rugosity of hard structures can reduce hydraulic loading.

More formally the following hypotheses are tested:

1. Wave velocity and height will reduce in the presence of species mimics to determine.
2. Wave velocity reduction will vary between species mimics, with more flexible species mimics reducing wave energy more effectively than stiffer species mimics.
3. Wave velocity and height will reduce in the presence of enhancement units.
4. Wave velocity reduction will vary between tile unit designs, with more complex/ increased surface texture of enhancement units will reducing wave energy more effectively than smoother, less complex surfaces.

5.2 Methodology

5.2.1 Experimental set-up

The wave flume at the University of Southampton (see Figures 5.1 & 5.2), is constructed from perspex, measuring 15.00 m long, 0.46 m wide and 0.46 m deep. The flume consisted of a piston type wave generator (non-wave absorbing) in the first 2.00 m of the flume and produced simple regular gravity waves. The flume design had a 1:10 perspex slope situated at 7.50 m distance from the wave generator with a height of 0.18 m and length 1.50 m. The slope plateaued to a raised perspex floor followed by dense foam blocks with a length of 2.00 m (heights 0.18 m) to simulate a sloping shoreline. A wave dampener, made of sponge material, was placed at 11.00 m, measuring 0.75 m in length to absorb the waves and reduce any possible reflection, which can distort the results. The last 1.25 m of the flume was left empty to monitor any wave movement and potential reflection occurring. The test section was 0.90 m in length and the width of the flume and was situated at 9.25 m and consisted of different textured test mats (details in Section 5.2.3). Water depth was filled to 0.28 cm and held constant for standing water.

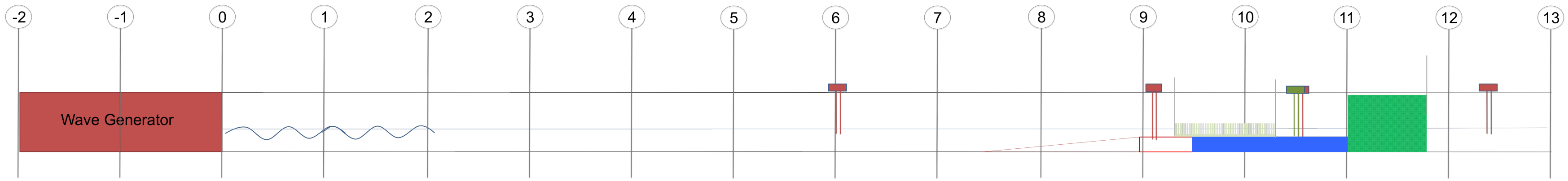

Figure 5.1: Representative full side view of experimental set up of wave flume at the University of Southampton.

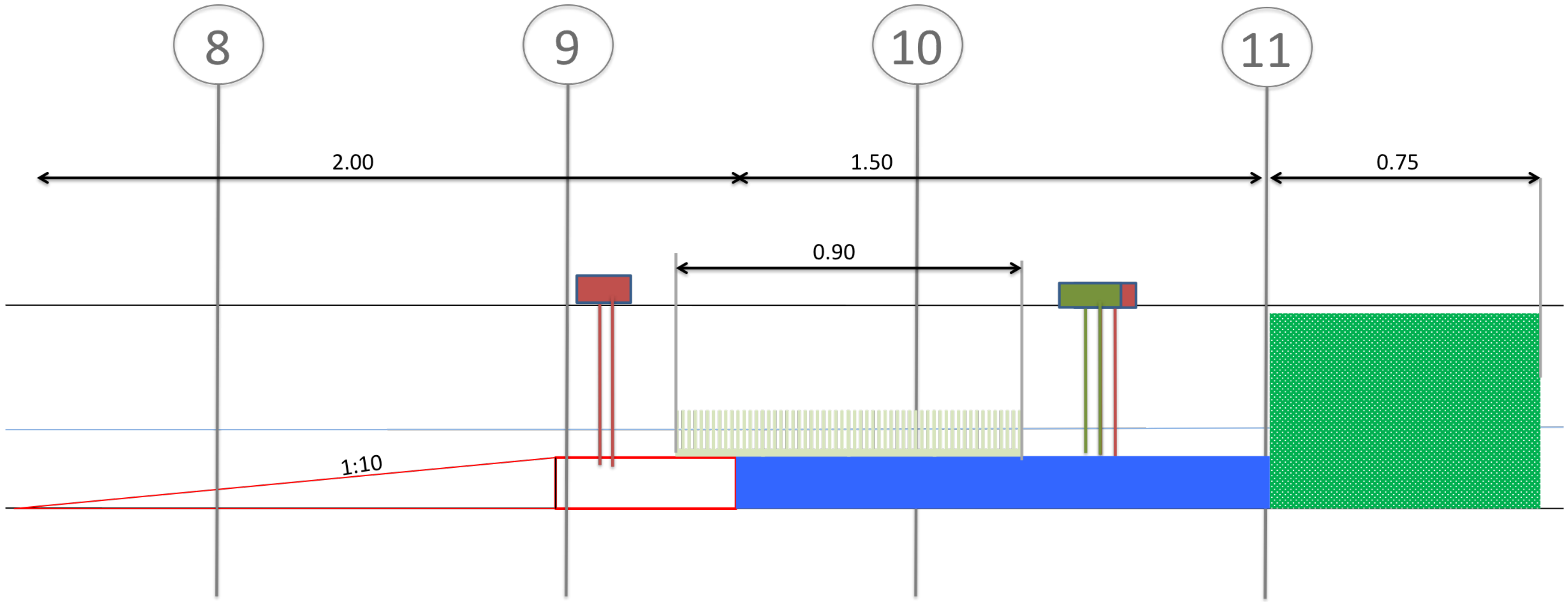

Figure 5.2: Zoomed side view of the experimental set up from 7.00 to 12.00 m of the test section in the wave flume. Small red full boxes represent wave probes, small green full box represents the Acoustic Doppler Velocimeter (ADV). Mimics and tile testing area shown in light green measuring 0.90 m. Dark blue rectangle represents foam floor, followed by dark green sponge material square. Slope outlined in red showing 1 in 10 slope angle.

5.2.1.1 Recording Equipment

A Nortek AS VectrinoPlus Acoustic Doppler Velocimeter (ADV) device was used to measure 3D flow velocities, situated behind the testing area. ADV devices record accurate measurements of small scale velocity fluctuations and are well recognised in eco-hydraulic studies (Paul et al., 2012). The ADV consists of three focused beams which measure the reflection of the acoustic signal off neutrally buoyant glass powder scattering particles (added for experiments) in the flow at high sampling rates (Wahl, 2000). Flow velocity is determined by calculating the magnitude of the phase difference from the acoustic echoes (Wahl, 2000).

Four twin wire wave probes (conductivity gauges) were deployed vertically in the centre of the flume (away from the turbulent effects caused by the flume wall) to measure wave height at different points in the flume. Gauges were placed at (1) 5.00 m, (2) 9.25 m, (3) 10.50 m and (4) 12.50 m from the wave generator in the middle of the flume. Probe *3* was the main wave height recording unit to determine wave height differences, whereas the other three were deployed to monitor wave conditions within the flume to avoid issues such as reflection.

Prior to data collection, calibration of the ADV settings and wave probes was carried out. Calibration of the ADV took place by evaluating two parameters, Signal to Noise Ratio (SNR) and Correlation Scores (COR), and configuring the optimum recording settings. Recordings are only accepted for SNR values of $>$15 dB and a COR of $>$70% (Wahl, 2000; Frostick et al., 2014). Wave probes were calibrated by recording voltage outputs at 0.01 m increments to a maximum water depth of 0.38 m, and then decreasing back to 0.28 m. Linear equations from the calibration chart (Figure 5.3) enabled calculations of the associated water level from the voltage outputs.

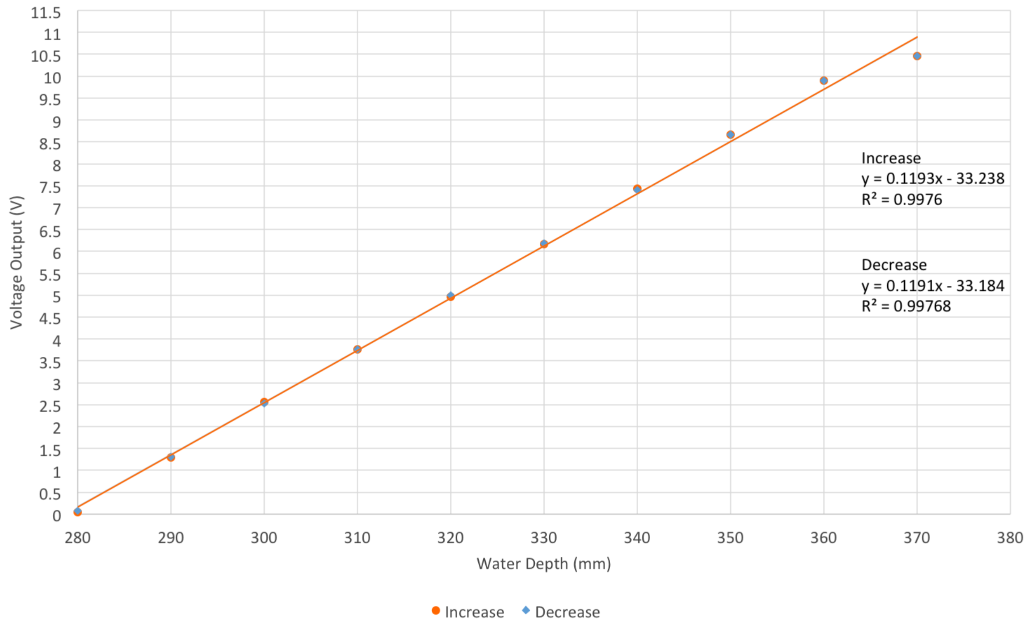

Figure 5.3: Calibration of the wave probes prior to experiments showing measurements as water levels are increased and decreased with water depth on the 'X' axis, and voltage output on the 'Y' axis. A line of best fit was added to each probe, and the R squared values, and linear equations calculated for use in the experiments to determine the water levels.

5.2.2 Data collection

Experiments were carried out at three different ampere's (electrical current generated to power the wave paddle) producing different wave heights. Wave height is related to the power and velocity of a wave and was therefore deemed a suitable control factor. In addition, the third wave height selected produced a breaking wave where the samples are located. This was included as part of the experiment to determine the wave attenuating capabilities of the mimics and tile designs under varying conditions, as would be seen in the natural environment.

0.75 Amps = 27 mm wave height (non-breaking waves)
1.0 Amps = 44 mm wave height (non-breaking waves)
1.5 Amps = 56 mm wave height (breaking waves)

A total of 20 x 30 second repetitions at each ampere and a frequency of 1 Hz were carried out recording wave heights and velocities passing over different mimic species and tile units (including a control for each) (see Table 5.1 & 5.2). Recordings for 30 seconds were selected to avoid any potential interference due to wave reflection which could develop during longer wave generation in a small tank. The ADV was configured to the following settings on the Vectrino Plus programme:

Sampling rate	50 Hz
Nominal velocity	±0.10 m/s
Transmission length	2.4 mm
Sample volume	9.1 mm

5.2.3 Test samples

Experiment 1: Mimics

The first set of experiments used eight mimic species of different levels of rugosity (based on flexibility and surface area) in the flume, which could be associated with different subtidal and intertidal species traits (Table 5.1). Different designs were selected with distinct characteristics in order to clearly define between species engineering properties.

Table 5.1: Dimensions (mm) of mimic designs used in the hydraulic experiments, their biological mimic representative and a qualitative assessment of flexibility where 'stiff' = none/ very little movement, 'moderate' = some movement but generally remains structured up right, and 'high' = full flexibility to move in any direction.

Design	Length	Width	Height	Mimic	Flexibility
Control	-	-	-	-	-
Sequins	5	5	2	Barnacles	Stiff
Studs	11	11	4	Limpets	Stiff
Bubble wrap (Full)	10	2	100	Kelp	High
Bubble wrap (Half)	10	2	50	Seaweed	High
Bubble wrap (Short)	10	2	10	Seaweed	High
Mussels	10	10	30	Mussels	Stiff
Artificial seaweed	30	30	160	Seaweed	Moderate
Artificial sea grass	10	2	50	Sea grass	Moderate

Experiment 2: Tiles

The second set of experiments used tiles with six difference surface designs to determine the effect of surface rugosity on wave velocity. The tile designs measured approximately 0.30 m length, 0.30 m in width and 0.03-0.05 m in depth (depending on design) (Table 5.2 & Figure 5.4). Designs were created to replicate simple micro-habitats which may exist on rocky shores, and basic enhancement methods adopted in previous research studies (see Chapter 4.)

Table 5.2: Dimensions of tile unit designs used in the hydraulic experiments. See Figure 5.4.

Design	Length (mm)	Width (mm)	Height (mm)
Control	300	300	30
Rough (Vertical)	300	300	30
Rough (Horizontal)	300	300	30
Holes	300	300	30
Squares	300	300	50
Wiggles	300	300	50

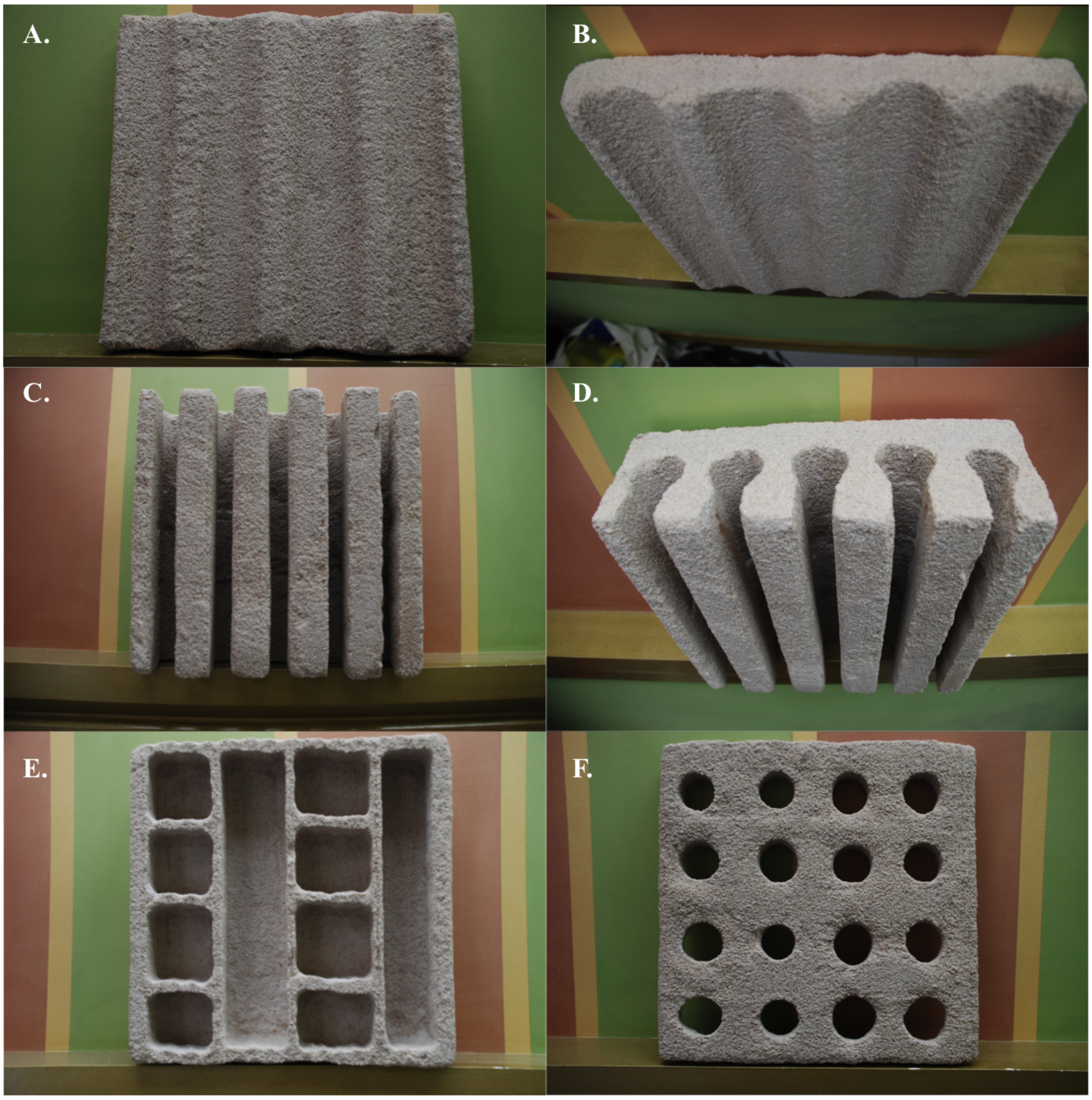

Figure 5.4: 3D printed tile units with five different designs of varying complexity. A & B = *Rough* design, C & D = *Wiggles*, E = *Squares*, F = *Circles*. Image supplied by Monolite UK Ltd.

5.2.4 Data Analysis

Signal to Noise Ratio (SNR) and Correlation Scores (COR) filtered Acoustic Doppler Velocimeter (ADV) data were analysed using WinADV software to calculate the Magnitude of the Average Vx, Vy and Vz Velocities (Mag V-Avg), where Vx = velocity on the X axis, Vy = velocity on the Y axis, Vz = velocity on the Z axis (Equation 5.1). Further calculation determined the reduction in wave velocities due to the presence of mimic species and tile units.

$$MagV - Avg = \sqrt{Avg - Vx^2 + Avg - Vy^2 + Avg - Vz^2} \tag{5.1}$$

Voltage outputs from the recorded data were converted to the associated wave heights using the linear equations calculated from the previous calibrations (Figure 5.3). Once wave heights were calculated, the difference in the average wave heights between the control and each test sample were calculating by determining reduction in wave height from the control and the percentage reduction.

To test our first and third hypotheses, Analysis of variance (ANOVA) was carried out to determine any significant differences in wave velocity as a result of wave heights and samples factors (mimics and tile units). Prior to ANOVA analysis, homogeneity of variance was confirmed using Levenes test. A two-way crossed design was used for both 'Experiment 1: Mimics' and 'Experiment 2: Tiles', each with fixed factors for Wave Heights (three levels, 27mm, 44mm, 56mm) and the respective samples: 'Experiment 1: Mimics' (nine levels: control, sequins, studs, bubble wrap (full), bubble wrap (half), bubble wrap (short), Mussels, artificial seaweed and aritifical sea grass); and 'Experiment 2: Tiles' (six levels: control, rough (V), rough (H), holes, squares and wiggles). To test our second and fourth hypotheses, post-hoc analysis of the two-way ANOVA using Tukey HSD tested for within factor differences between factors and levels.

5.3 Results

5.3.1 Experiment 1: Mimics

Eight variations of intertidal and subtidal coastal species were selected to test the impacts of different physical properties on wave energy. Wave flume experiments showed a reduction in wave velocity across all mimics compared to the control experiment (Figures 5.5 & 5.6). The lowest wave height (27 mm waves) showed the greatest reduction in wave energy across all mimics followed by the largest wave height (56 mm waves). The full length bubble wrap, representing a kelp type property, showed the highest percentage reduction at all wave amperes. Two mimics with very different properties representing barnacles (studs) and small seaweed type features, showed the least reductions overall.

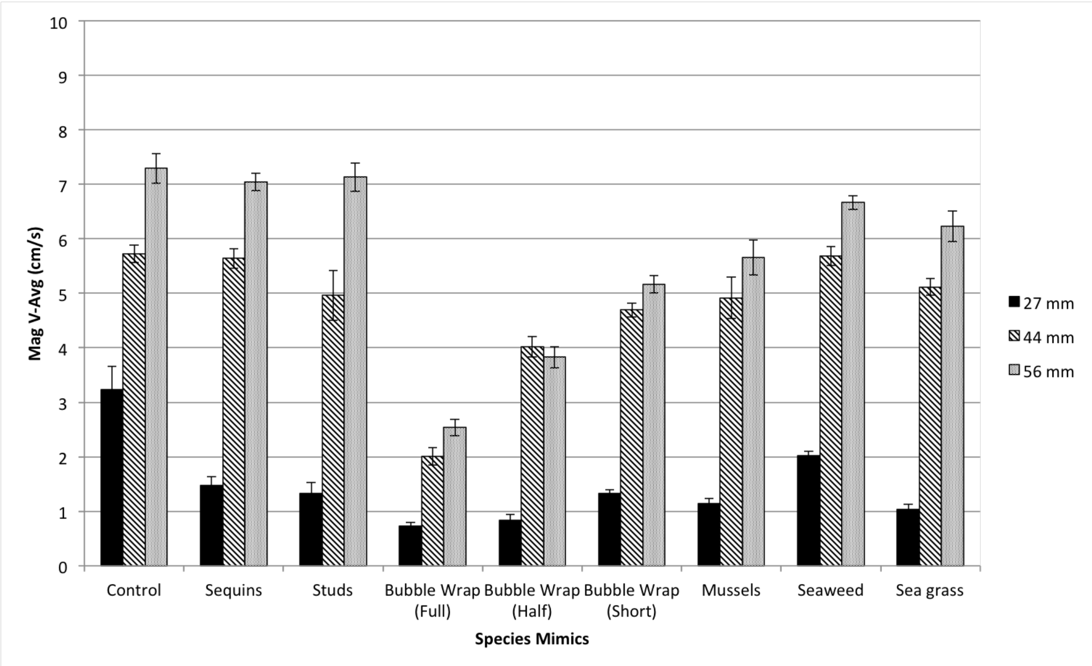

Figure 5.5: Average wave velocity (± standard error) recorded after travelling over each mimic species, recorded at three different wave heights.

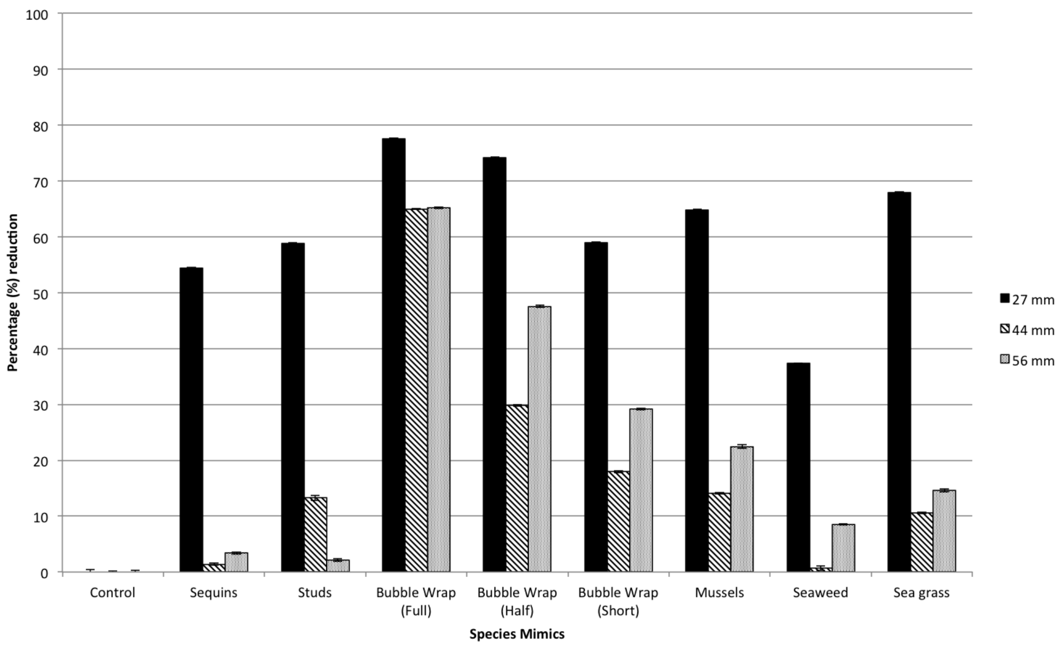

Figure 5.6: Mean percentage wave velocity reduced (± standard error) after travelling over each mimic species, recorded at three different wave heights.

Results of the two-way ANOVA test (Table 5.3) showed that there was a significant difference in wave velocity between all mimics and at different wave height conditions. Moreover, there was a significant difference in the effect on wave velocity due to the interactions between the two factors, wave heights and species mimics. Post-hoc analysis of the two-way ANOVA using Tukey HSD showed significant differences between some mimic species across all three wave heights. Closer inspection of the interactions based on differences between species mimics split across the three wave heights showed mostly no significant difference between species mimics at the 27 mm wave height (10 significantly differences out of 36 between species comparisons), but the number of significantly differences increased with wave height (44 mm = 11 out of 36 comparisons; 56 mm = 21 out of 36 comparisons). This could be due to the breaking waves which create additional reduction in wave energy but may be assisted further but the presence of mimics. Significant differences in wave velocities at the lowest wave height (27 mm) were mostly between all mimic samples and the control. This is likely due to the lack of energy created by the waves at such a low height, therefore any resistance was significant.

The species mimics with the highest number of non-significant differences (>0.05) with other mimics across all three wave heights were the sequins, studs and sea grass. Half and short length bubble wrap showed little significant difference between them indicating they have similar wave reducing properties, whilst the full length bubble wrap showed the most significantly difference against other mimic species. The bubble wrap mimics, used to represent a long seaweed or kelp species, displayed the highest levels of wave velocity reduction across all three different test lengths and wave heights, but in particular showed significant differences in their wave energy reducing ability compared to most other species mimics at higher wave heights. Other species which did not significantly differ from the bubble wrap mimics were the seaweed and sea grass mimics, which displayed moderate flexibility and occupancy within the water column. The differences in the bubble wrap compared to other species mimics (with stiffer properties) and not the seaweed and sea grass, is likely due to the higher flexibility and a larger presence in the water column. Other species such as sequins and studs species, which would likely represent barnacles and limpets, have stiff properties, were notably small and displayed the lowest wave attenuation of all the mimics. Mussels also displayed stiff properties, however, they provided a considerable presence in the water column. This indicates that size also plays an important role in reducing wave velocity, as would logically be expected.

Table 5.3: Results from a Two-Way ANOVA analysis of the hydraulic experiments to determine if wave heights and different species mimics do have a significant impact on wave velocity.

Factor	Sums of Square	df	Mean Square	F	P
Wave Height (WH)	1030.380	2	515.190	1069.372	<0.001
Species Samples	294.578	8	36.822	76.431	<0.001
WH*Samples	88.245	16	5.515	11.448	<0.001

Table 5.4: Results from a Tukey HSD post-hoc analysis of the hydraulic experiments to determine differences in the species mimic effects on wave velocity. BW = Bubble Wrap. *Italicised* values denote a significant difference.

27 mm Wave Height

	Control	Sequins	Studs	BW (Full)	BW (Half)	BW (Short)	Mussels	Seaweed	Sea grass
Control									
Sequins	*<0.001*								
Studs	*<0.001*	0.999							
BW (Full)	*<0.001*	0.035	0.348						
BW (Half)	*<0.001*	0.099	0.591	1.000					
BW (Short)	*<0.001*	0.998	1.000	0.230	0.460				
Mussels	*<0.001*	0.664	0.994	0.681	0.909	0.989			
Seaweed	*<0.001*	0.286	0.183	*<0.001*	*<0.001*	0.087	*0.005*		
Sea grass	*<0.001*	0.595	0.969	0.959	0.997	0.951	1.000	0.008	

44 mm Wave Height

	Control	Sequins	Studs	BW (Full)	BW (Half)	BW (Short)	Mussels	Seaweed	Sea grass
Control									
Sequins	1.000								
Studs	0.251	0.439							
BW (Full)	*<0.001*	*<0.001*	*<0.001*						
BW (Half)	*<0.001*	*<0.001*	0.169	*<0.001*					
BW (Short)	0.029	0.076	0.998	*<0.001*	0.603				
Mussels	0.183	0.346	1.000	*<0.001*	0.224	0.999			
Seaweed	1.000	1.000	0.519	*<0.001*	*<0.001*	0.128	0.430		
Sea grass	0.558	0.763	1.000	*<0.001*	0.059	0.955	1.000	0.800	

56 mm Wave Height

	Control	Sequins	Studs	BW (Full)	BW (Half)	BW (Short)	Mussels	Seaweed	Sea grass
Control									
Sequins	0.988								
Studs	1.000	1.000							
BW (Full)	*<0.001*	*<0.001*	*<0.001*						
BW (Half)	*<0.001*	*<0.001*	*<0.001*	*0.001*					
BW (Short)	*<0.001*	*<0.001*	*<0.001*	*<0.001*	*0.003*				
Mussels	*<0.001*	*<0.001*	*0.001*	*<0.001*	*<0.001*	0.831			
Seaweed	0.440	0.922	0.901	*<0.001*	*<0.001*	*<0.001*	0.047		
Sea grass	0.011	0.112	0.165	*<0.001*	*<0.001*	0.030	0.674	0.900	

5.3.2 Experiment 2: Tiles

Six different tile unit designs were tested to determine whether surface complexity can impact wave velocity, and therefore energy in an intertidal environment. A plain tile unit with no surface design implemented was tested as a control to compare to the other five designs of varying complexity. Wave flume experiments showed a reduction in wave velocity across all tile designs compared to the control experiment (Figures 5.7 & 5.8). Reduction in the wave velocity was greatest at the lowest wave height (27 mm waves) which showed the greatest dissipation across all tile designs, similar to the species mimics. This was followed by the second largest height (44 mm) and the least reduction occurring at the greatest height (56 mm). The rough tile units displayed the most notable differences between wave reduction. These samples were from the same tile unit but using different orientations against the waves. The results from these two orientations show strong differences between wave reductions at all wave heights, with the horizontal orientation creating rugosity features perpendicular to the wave flow, having a greater impact on wave energy reduction compared to the vertical design which has rugosity features running in the direction of wave flow. The 'square' and 'wiggle' designs showed high dissipation, particularly at the larger wave heights, compared to the other designs which is most likely due to the increased height in the tile units.

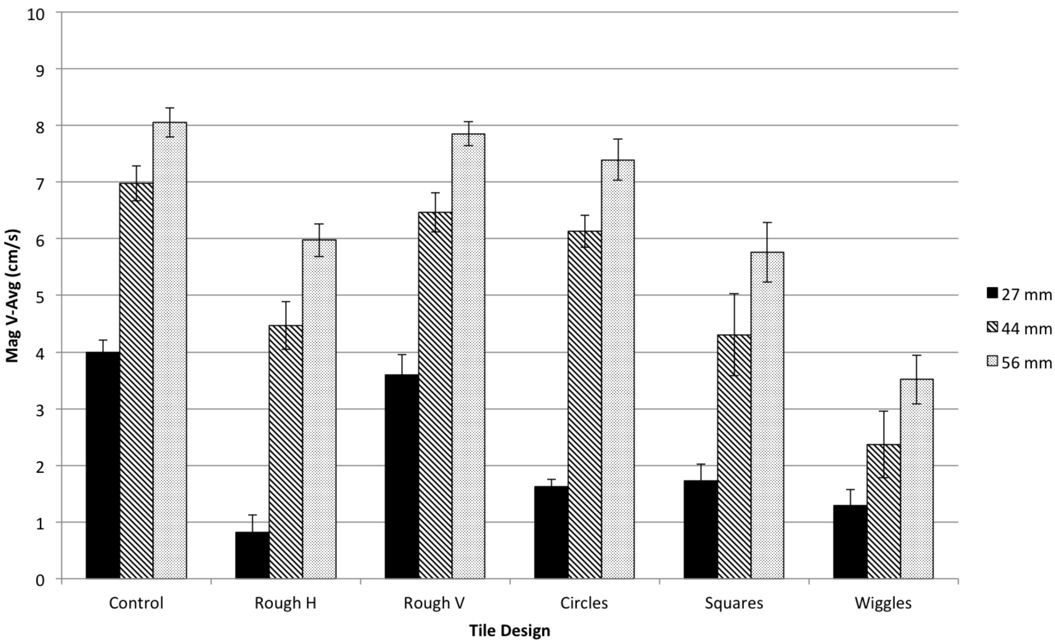

Figure 5.7: Average wave velocity (± standard error) recorded after travelling over each tile unit, recorded at three different wave heights.

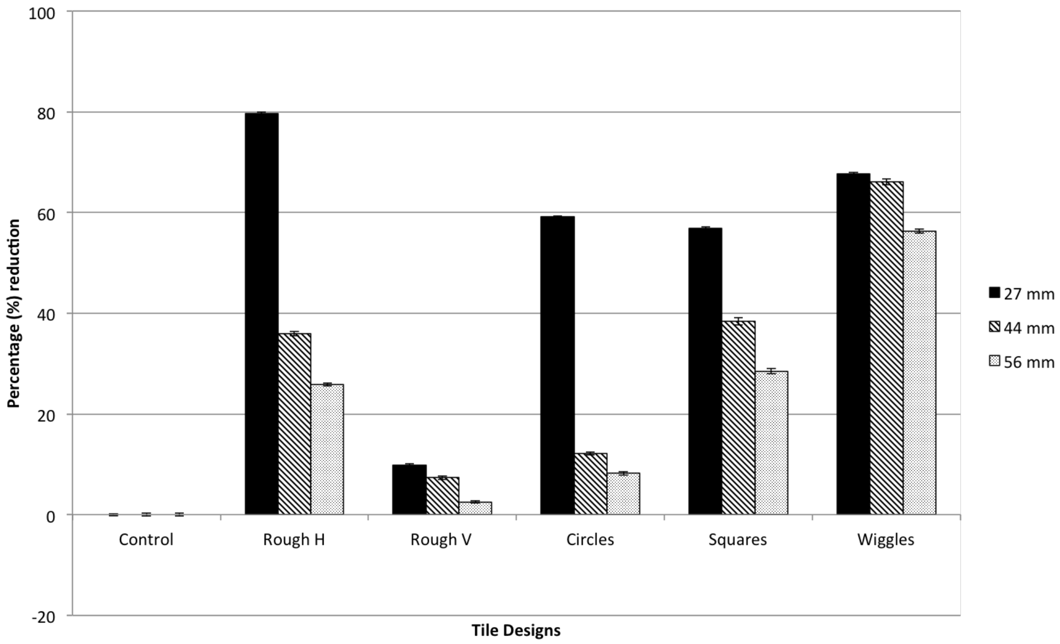

Figure 5.8: Mean percentage wave velocity reduced (± standard error) after travelling over each tile unit, recorded at three different wave heights.

Results of the two-way ANOVA test (Table 5.5) showed that there was a significant difference in wave velocity between all tile designs and at different wave height conditions. Moreover, there was a significant difference in the effect on wave velocity due to the interactions between the two factors, wave heights and tile unit design. Post-hoc analysis of the two-way ANOVA using Tukey HSD showed significant differences between most tile unit designs, particularly at the lowest wave height (27 mm) and largest wave height (56 mm). Similar to 'Experiment 1: Mimics', at the lowest wave height (27 mm), all designs apart from the Rough vertically orientated tile (pattern parallel to wave flow) showed significant differences in wave velocity to the control. This is likely due to the low energy energy waves at the lower height being more susceptible to dissipation from resistance. 'Wiggle' design showed significant difference compared to most other designs across all three wave heights. This is likely due to it's larger height (50 mm) compared to most others (30 mm). Of particular interest, the rough vertically orientated design showed no significant difference in comparison to the control unit across all three wave heights (p >0.05 for all three heights). The Square unit designs also showed no significant differences across all three wave heights compared to the rough horizontally orientated design and circles. This could be due to the designs creating similar dissipating qualities.

Table 5.5: Results from Two-Way ANOVA analysis of the hydraulic experiments to determine if tile design does have a significant impact on wave velocity.

Factor	F	P
Wave Height (WH)	202.948	<0.001
Tile Samples	47.764	<0.001
WH*Samples	3.581	<0.001

Table 5.6: Results from a Tukey HSD post-hoc analysis of the hydraulic experiments to determine differences in the tile unit design effects on wave velocity. *Italicised* values denote a significant difference.

27 mm Wave Height						
	Control	Rough H	Rough V	Circles	Squares	Wiggles
Control						
Rough H	*<0.001*					
Rough V	0.926	*<0.001*				
Circles	*<0.001*	0.164	*<0.001*			
Squares	*<0.001*	0.107	*<0.001*	1.000		
Wiggles	*<0.001*	0.712	*<0.001*	0.914	0.811	

44 mm Wave Height						
	Control	Rough H	Rough V	Circles	Squares	Wiggles
Control						
Rough H	0.011					
Rough V	0.972	0.062				
Circles	0.804	0.178	0.995			
Squares	*0.003*	1.000	0.02	0.075		
Wiggles	*<0.001*	0.051	*<0.001*	*<0.001*	0.061	

56 mm Wave Height						
	Control	Rough H	Rough V	Circles	Squares	Wiggles
Control						
Rough H	*0.004*					
Rough V	0.999	0.011				
Circles	0.797	0.087	0.949			
Squares	*0.001*	0.998	*0.003*	0.025		
Wiggles	*<0.001*	*<0.001*	*<0.001*	*<0.001*	*0.001*	

5.4 Discussion

The results confirm all of the hypothesis and show significant differences between 1) presence and absence of mimic species; 2) between mimic species; 3) presence and absence of designed tile units compared to the control; 4) between tile unit surface designs.

5.4.1 Experiment 1: Mimics

Of the eight variations of intertidal and subtidal coastal species mimics that were selected to test, there were significant differences in the reduction in wave velocity over the three wave heights. The results showed wave height was a significant factor contributing to wave velocity. Overall, all species reduced wave velocity, however, the largest reduction was seen at the lowest wave height (27 mm waves), and the second largest wave height (44 mm waves) displayed the least amount of wave velocity reduction overall. This is most likely due to the wave formation and breaking that occurred at the 56 mm waves, compared to the 44 mm waves. The wave generating settings were specifically selected to provide comparisons of wave velocity reducing effects on breaking and non-breaking waves over the mimics, and the results highlighted differences. Breaking waves create larger amounts of turbulence, therefore aiding in the reduction of wave velocity at the highest wave height (Svendsen, 1987). This suggests the presence of any mimic is effective at low wave height levels, however, the higher the wave height, the more specific species are required in order to reduce wave energy significantly.

In addition, the interaction between wave height and species mimics showed significant results. Closer inspection of the statistical analysis for the interaction between wave height and species mimics showed wave velocity was significantly different between all species mimics and the control, particularly at the 27 mm wave height, but also at the 56 mm wave height (across most of the mimics), but varied within species at different heights. Further observations noted that more flexible species showed significant differences in wave reduction compared to stiffer species. Moreover it was established that longer, more buoyant and flexible species (such as kelps, seaweeds and sea grasses) show greater abilities to reduce wave velocity, particularly at larger wave heights, followed by larger but more rigid species. Larger species can occupy a greater volume of the water column, therefore creating more resistance. This indicates that area occupied in the water column and flexibility are the primary factors and species characteristics affecting wave velocity. Augustin et al. (2009) found that a larger area of the water column was a differentiating factor in the ability to attenuate waves for wetland vegetation. Overall, this study shows the presence of any species can provide some form of resistance and wave attenuating property; however, the effectiveness of species to reduce waves depends on their physical properties and the wave conditions.

Despite the conclusion that longer and more flexible species are most effective at reducing wave energy, particularly at larger wave heights; the physical stress of the environmental pressures on the species and their ability to withstand greater energies must also be considered. More flexible species are potentially more fragile and less hardy, and therefore cannot withstand high physical stresses for prolonged periods. Several different species taxonomies have been studied and shown to be effective at wave attenuation (see Appendix B). Plant species such as various types of sea grass and macrophyte species (e.g. *Spartina alterniflora, Phragmites australia, Salicornia* spp., have been extensively studied and demonstrated excellent wave attenuating properties within coastal environments through a number laboratory and field studies (Möller, 2006; Möller et al., 2011; Ysebaert et al., 2011b; Möller et al., 2014; Anderson and Smith, 2014) *Zostera marina*). These types of species are found globally, regularly occupying intertidal areas, and are therefore potentially suitable species for transplanting and using in soft coastal engineering schemes to facilitate wave reduction in the coastal zones. However, studies have shown that wave velocity and wave breaking can impact the structure and function of some species over time by creating damage (Möller et al., 2014; Carus et al., 2016). For this reason, macrophyte species may be less suited to high energy environments, despite their generally flexible physical properties.

Species such as bivalve (e.g. mussels and oysters) are likely to be more suitable in stronger wave conditions due to their tougher shell exterior and development of byssus threads used to attach themselves to a variety of substrates. Bivalve species have been used in several laboratory and field studies looking at wave energy reduction, and shown to be effective at wave attenuation (Borsje et al., 2011; Scyphers et al., 2011; Donker et al., 2012; Manis et al., 2015) (see Appendix B for more details on studies). Witman and Suchanek (1984) and Donker et al. (2012) noted that mussel byssus threads adapt to wave forcing and exposure, which is likely why they can survive in high wave exposed environments (Waite, 1983; Waiter, 1999; Bell and Gosline, 1996; Hunt and Scheibling, 2001). Moreover, mussels have also known to host diverse fauna communities (Hewatt, 1935; Dittmann, 1990; Hammond and Griffiths, 2004), providing protection from predators (Thiel and Ullrich, 2002) and trapping larvae (Gutierrez et al., 2003). As intertidal species, they can also buffer high temperatures at low tide (Stephens and Bertness, 1991) and can improve water quality through their filtration systems (Borsje et al., 2011). They are also able to survive fluctuating environments and withstand low tidal levels compared to most vegetative species who are better evolved for subtidal environments or lower exposure levels. These are important natural qualities that should be considered in future for further studies or potential implementation of species within coastal engineering schemes. Furthermore, species reactions to temperature, water nutrient content, current flows and other physical stresses are essential factors that will impact the potential survival and success of species within engineering schemes. Introduction of species to new areas should carefully consider the environmental conditions and compare them to the biological requirements of potential species in order to increase effectiveness.

5.4.2 Experiment 2: Tiles

To date, a number of ecologically enhancing features have been designed, implemented and studied, with very successful outcomes (Moschella et al., 2005; Burt et al., 2009b; Coombes et al., 2011; Firth et al., 2012, 2014; Evans et al., 2016). Incorporation of such techniques has increased biodiversity and abundance of species on artificial surfaces containing these features, compared to artificial surfaces without (Chapman and Blockley, 2009; Martins et al., 2010; Browne and Chapman, 2011; Firth et al., 2012; Perkol-Finkel and Sella, 2015; Ido and Shimrit, 2015; Evans et al., 2016). While the design of these ecological features is to increase structural complexity and habitat heterogeneity on artificial structures, thus promoting biodiversity, there has been no consideration of their physical presence and the multifunctional role they may provide. The results showed that increased surface complexity successfully reduced wave velocity by up to 80% at low wave heights, and approximately 30-40% at higher wave heights (Figure 5.6). Furthermore, this study demonstrated that the type of structural complexity significantly contributes to the levels of wave attenuation (Table 5.5).

Square and wiggle designs showed the highest levels of wave velocity reduction overall, however this is to be expected when considering their larger size compared to the other three. Comparison between the rough (horizontal and vertical orientations) and the circular designs, showed interesting results. Horizontal placement of the rough textured tiles displayed the highest levels of wave attenuation, followed by the circular design and the vertical rough tile. The rough vertical design also showed no significant difference to the control design across all three wave heights, indicating it provides very little resistance or benefit. Between factor interactions showed similar results to the species mimic experiments, that resistance of most type reduces wave velocities at low levels. However, larger features seem to play a significant difference in terms of enhancement features and wave velocity reduction. These results highlighted that not only does structural complexity and design play a significant role in the reduction of wave velocities, orientation of designs is also a fundamental factor. The rough vertical and horizontal design experiments were investigated using the same tile with different orientation within the wave flume. Structural designs which provide roughened surfaces and are perpendicular to the flow of the waves, reduce wave velocities by over three times that of the same design but orientated within the flow of the waves.

This study highlights the importance of structural surface complexity, not only for promoting biodiversity and species richness (Chapman and Blockley, 2009; Martins et al., 2010; Browne and Chapman, 2011; Firth et al., 2012; Perkol-Finkel and Sella, 2015; Ido and Shimrit, 2015; Evans et al., 2016), but also fulfilling an engineering role through reducing wave velocity. This physical engineering role of enhancement features has not been considered previously, and could add even more value towards new environmentally sensitive coastal defence schemes. Future studies should include field measurements of wave velocities and loading on the coastal area with and without the presence of enhancement features, using sensor pads or tidals gauges fixed on the structures to determine the extent of wave energy reduction in a full-scale environment. Furthermore, investigations considering the location and orientation of such enhancement features would provide valuable supplementary evidence to support this study.

5.5 Conclusions

The use of ecosystem engineering species has been studied in several numerical, laboratory and some field experiments. However, research to date has been limited to a small number of species and has not considered the variation in species properties and their effects of wave energy. There is also a gap looking at their multifunctional roles in engineering and providing increased habitat space for other intertidal and subtidal species. In addition, while hard structural enhancement features are designed to promote biodiversity on artificial structures, there has been no consideration of their engineering role in reducing wave energy. This study carried out physical wave studies, investigating the reduction of wave velocities due to the presence of eight different ecosystem engineering mimics, and five different structural surface designs. The results showed significant differences between all of the mimic species, with all of them positively reducing wave velocities. There was also a significant reduction in wave velocities due to increased surface complexity. This study shows that the presence of species or any added complexity, can successfully reduce wave velocities, however closer consideration to specific species and design features must be made for increasing wave height conditions. Furthermore it demonstrates the multifunctional role of ecosystem engineers and structural enhancement units by increasing habitat availability and reducing wave velocities while increasing biodiversity. Although two different engineering methods, with very separate primary roles, both show successful capabilities in biological and engineering roles.

Chapter 6

General Discussion

6.1 Thesis overview and summary

Predicted sea level rise and increasing storminess (Marshall, 2001; IPCC, 2014; Nicholls et al., 2014, 2015) are contributing to the need for continued developments of hard coastal protection globally (Govarets and Lauwaert, 2009; Young et al., 2011). The environmental repercussions through loss and modifications of natural coastal habitats have motivated research on ecologically sensitive and multifunctional designs of coastal defence structures (CDS), also providing incentives to develop and evolve current marine planning policies (UK Parliament, 2011; Naylor et al., 2012).

Hard artificial structures are generally regarded as poor habitat substitutes for natural rocky shores due to the high levels of environmental disturbance and lack of structural complexity (Chapman, 2003; Moschella et al., 2005; Carvalho et al., 2013; Firth et al., 2013b; Aguilera et al., 2014; Evans et al., 2016; Firth et al., 2016), leading to lower diversities of species and relative abundance of taxa (Connell and Glasby, 1999; Chapman, 2003; Knott et al., 2004; Bulleri et al., 2005; Pinn et al., 2005; Pister, 2009; Burt et al., 2010, 2011). The need for enhancement of artificial structures for secondary functions including boosting biodiversity or other ecosystem services has become more recognised, particularly over the past decade (Chapman and Blockley, 2009; Chapman and Underwood, 2011; Firth et al., 2014; Perkol-Finkel and Sella, 2015; Evans et al., 2016). Although the concept of ecological engineering is not new (Schulze, 1996; Bergen et al., 2001), it is still an emerging and relatively untested approach in terms of intertidal marine environments. Successful research in marine ecological enhancements has identified potentially suitable methods of improving habitat heterogeneity on artificial CDS (Chapman and Blockley, 2009; Browne and Chapman, 2011; Chapman and Underwood, 2011; Firth et al., 2014; Ido and Shimrit, 2015; Perkol-Finkel and Sella, 2015; Evans et al., 2016; Firth et al., 2016). This concept however, still lacks practical examples and evidence of effective implementation (but see Harris, 2003; Scyphers et al., 2015). Knowledge gaps and the current levels of understanding between engineering demands and designs, and ecological requirements is limiting our capabilities to provide guidance on best practice designs.

My thesis investigated the role of artificial CDS as suitable habitats for rocky shore biodiversity, and the potential of novel hard and soft engineering designs, materials and methods to improve multifunctional coastal protection. Four key areas were identified as essential knowledge gaps to address in order to (i) improve our understanding of the interactions between intertidal ecology and hard coastal protection, (ii) effectively incorporate ecologically-sensitive and multifunctional design into CDS. In this thesis I investigated:

1. the difference in biological communities and topographic complexity between natural and artificial shores;
2. the capability of porous CDS to provide functional habitat spaces for coastal assemblages via their internal environment;
3. the potential for 3D printing as an effective material and method for adding complexity and ecological value to CDS;
4. the capability of marine species and structural enhancing features to reduce hydraulic loading on coastlines.

In this concluding chapter, I summarise the key findings from this research and their applicability to coastal engineering and management. I then discuss some of the challenges experienced and identified through this research which can be overcome in future studies and projects. Finally, I discuss the practical applications and knowledge gaps remaining in order to progress effective implementation of ecological engineering practices within industry and management schemes.

6.1.1 Engineering for Ecology

Previous studies report that coastal biological communities supported by artificial structures are different and less diverse than those on adjacent natural rocky shores (Chapman, 2003; Chapman and Bulleri, 2003; Pinn et al., 2005; Moschella et al., 2005; Pister, 2009; Firth et al., 2013c; Aguilera et al., 2014; Firth et al., 2016). Furthermore, Moschella et al. (2005) suggest that habitat spatial scale plays a role in the colonisation of coastal species, and enhancement at the micro-scale level is most successful for encouraging colonisation. Other studies show effective implementation at larger scale-levels (Burt et al., 2009b; Coombes et al., 2011; Firth et al., 2012, 2014; Evans et al., 2016). The effects of implementing modifications to structures at different spatial scales remains unclear. In Chapter 2, I surveyed seven sites on the south coast of England, four natural and three artificial. I investigated whether there was a significant difference between biodiversity and topography on natural and artificial shores, and if so, whether scale level was a key factor in determining any differences. The results of this study showed there was a significant difference in biological communities between natural and artificial shores, and the seven different sites independently. Furthermore, I found that characterising species differed between natural and artificial shores. Natural shores tend to be characterised by species such as fucoids and some foliose red algae, while artificial shore communities are largely characterised by invertebrate species such as limpets and barnacles. These results suggest hardy and more adaptive species which can retreat to sheltered areas are more likely to successfully survive on artificial shores. In addition, I investigated the relationship between topographic complexity and biological communities, for which I found no significant relationship. However, quantitative assessment of micro-habitat areas was not carried out and should be considered in future experiments. Micro-habitats such as cracks, crevices and rock pools, provide essential refuge spaces to protect species during low tide or strong wave conditions (Chapman and Blockley, 2009; Carvalho et al., 2013), and are generally lacking on artificial shores compared to natural shores (Chapman, 2003; Moschella et al., 2005; Aguilera et al., 2014; Evans et al., 2016; Hawkins et al., 2016). However, observations from my study, I found differences between micro-habitat types and the abundance across all sites, even between natural shores. Therefore, further work should be carried out investigating the abundance and types of micro-habitats present across a range of natural and artificial shores.

Following Chapter 2, I surveyed a porous rock armour groyne (aged approximately 20 years old) during engineering works to reduce its size (Chapter 3). The engineering works allowed me to access and record internal biological communities to compare with those on the external areas. To date, the internal areas of porous CDS have not been investigated or actively considered as potentially suitable habitats for coastal assemblages. The results of this survey showed significant differences between biological communities colonising the internal and external environments provided by the porous structure. Moreover, it showed that internal habitats support a higher species diversity (but not significantly different) compared to external environments with nearly twice as many species, particularly mobile species, recorded on internal as opposed to external surfaces. The placement of CDS in areas of higher environmental stresses influences coastal species compositions (Moschella et al., 2005; Burcharth et al., 2007; Vaselli et al., 2008; Pister, 2009; Firth et al., 2013b), particularly mobile invertebrate species that are able to retreat and select more suitable habitats. In addition, my study considered the additional benefits for coastal species that porous CDS provide in protection from routine maintenance works through internal shelter. I found significant levels of disturbance on units based on their placement on the structure. Anthropogenic disturbance is acknowledged as a severe form of ecological disturbance (Airoldi and Bulleri, 2011) and minimising the impacts of essential engineering works by providing sheltered habitat spaces through internal cavities within a structure should be more actively considered as part of coastal protection schemes.

Taken together, these studies (Chapters 2 & 3) showed that artificial shores have the potential to become suitable habitats for some species, and should be actively considered as such. They suggest that a key factor lacking on artificial shores, and resulting in lower species diversities, is sheltered habitat spaces, often found on natural rocky shores (Johnson et al., 2003). While previous research suggests that this could be provided by external micro-habitats (Moschella et al., 2005), my research showed sheltered areas of any capacity are valuable habitat spaces for coastal species. Porous structural designs serve as effective coastal protection through their wave dissipating properties (Dalrymple et al., 1991; Losada et al., 1995; Garcia et al., 2004; Burcharth et al., 2015) and, as shown in this research, can provide environmental benefits. Engineers should consider ways to incorporate more sheltered spaces within CDS designs, adding multi-functionality and allowing species to retreat or settle.

In Chapter 4 I investigated the use of an innovative and emerging technology that could be used for creating more complex ecological enhancement feature designs. *D-Shape* technology produces an environmentally neutral rock-like material (see Appendix 3 for details) with strength suitable for construction use (Lim et al., 2012; Jakupovic, 2013). While this technology and material have been applied to a number of markets, namely architecture (Bogue, 2013) and a number of artificial reefs (Dini, 2014; Boskalis, 2016), it has only just been considered or tested for use in coastal maritime engineering. I deployed and monitored the biological colonisation and physical durability of 44 *D-Shape* 3D printed units of five varying designs to determine if this material has the potential to be used in coastal engineering as an enhancement feature. Over a seven month period, 42 of the tiles showed extreme signs of weakness through erosion, cracking and dislodgement. The two remaining tiles did show signs of biological colonisation, particularly by *Ulva* spp.; however, species preference and the capability to promote biodiversity could not be determined. Nevertheless, this study provided useful information regarding the material, which at this prototyping stage is essential for the refinement of future designs and schemes. From this study, it is clear that the design specifications needs to be addressed before further consideration in coastal engineering. While the feature specifications of this material may be suitable for use in architecture (Lim et al., 2012; Bogue, 2013) and previous submerged artificial reefs (Dini, 2014; Boskalis, 2016), this study shows that the higher intensity environmental pressures of intertidal coastal areas requires the material to have more reinforcement. Larger scale prototypes should be used to the design specification limits in order to determine the true limitations for use in coastal engineering as most of the issues seemed to stem from the fragility of the small-scale designs. In addition, fixing methods for smaller "accessory" samples, similar to those used in this experiment, should be considered. The fixing method appeared to be a large factor contributing to the loss of the tiles by adding areas of susceptibility to erosion. The construction strengths (Jakupovic, 2013), design flexibility, use of different substrate types (Lim et al., 2012) and reduced environmental footprint, mean with adequate refinement *D-Shape* material could be a more suitable and effective tool for implementing ecological sensitive modifications to coastal protection. As a result of my research, further experimental designs and incorporation in future schemes has already been discussed and initiated with Arup and a number of their clients and projects. Many high-profile maritime projects (e.g. Thames Tideway West, Garden Bridge, Singapore Jurong Coastal Development and Swansea Bay Tidal Lagoon) have taken interest in the use of this technology and recognise it as a potentially market leading tool. Arup have also included this technology and research in their annual UKMEA Sustainability Report (Arup, 2016) which is an external facing document designed to showcase the most novel and innovative sustainability work being done to current and future clients.

6.1.2 Ecology for Engineering

The second overarching theme in this thesis was to look at the use of working with nature as effective and natural coastal protection methods. Globally, there are various ecosystems and ecosystem engineering species which have been established as successful natural coastal protection due to their ability to dissipate wave energy and reduce current velocities, while providing habitat areas for other coastal species (Gedan et al., 2010; Kamphuis, 2010; Temmerman et al., 2013; Carus et al., 2016). Previous research has recognised that the engineering properties of these natural coastal species are dependent on the characteristics of the species used (Mendez and Losada, 2004; Anderson et al., 2011). However, most studies have focused on the capabilities of specific species or comparisons between taxonomic order and have not directly compared the effects due to different species properties. Baring a few studies (Scyphers et al., 2011; de Vriend et al., 2014a; Manis et al., 2015), there is also a lack of evidence of effective practical implementation in coastal defence schemes. For successful incorporation into coastal protection schemes understanding of the impacts of the different physical properties and the biological demands is fundamental in order to competently advise engineers which species are the most appropriate for use in different schemes. In addition, while focus on ecologically sensitive and multifunctional CDS designs has been on the benefits to biodiversity, little consideration has been given to the engineering role enhancement of biology features may play.

In Chapter 5, I explored the physical capabilities of mimic marine species and different complexities of ecological enhancing features at reducing wave velocities. The first set of experiments compared eight different mimic species of varying physical properties to a control (no mimics present). The physical properties ranged in flexibility and presence (area covered) within the water column, and were tested over three different wave heights, including breaking and non-breaking waves. The results showed a significant difference in wave velocity reducing properties across all mimics and at each wave height. In particular those that were most flexible and with the largest area provided the greatest resistance and reduction in wave velocity, followed by more rigid species. Flexible species naturally follow the flow of the water, and particularly those which are longer, have the potential to create extended resistance over larger distances as they "go with the flow". However, rigid species showed good resistive properties but have a smaller surface area. Their rigid presence may have caused increased reflection and turbulence, therefore not reducing wave velocity as efficiently. Overall, it was clear that the presence of rougher and more textured surfaces creates increased resistance. Although logical, this has not been comparatively tested and provides new evidence for engineers and practitioners. Combined with biological requirements, it suggests ways for implementing more natural coastal resistance to defence schemes. This research shows that, although the use of species on their own may not be able to reduce wave loading fully on coastal areas, it has a significant impact and may be able to be combined with hard engineering structures to reduce structural sizes, increase life spans, minimise maintenance requirements and reduce beach nourishment programmes through sediment accretion and reducing sediment transport. Consideration still needs to be made as to the full scale capabilities of different species within real-life environments and the feasibility of species surviving and providing effective protection within high energy environments. Future studies need to be carried out with full-scale experiments representing realistic coastal conditions and actual species in order to fully compare and understand the hydraulic potential of different species.

The second part of this experiment examined the physical presence of tile samples with different surface complexities, representing different types of enhancement features. Overall, there was a significant reduction in wave velocity due to the presence of all of the different tile units compared to the control tile unit (no design), apart from the Rough vertically orientated design, which had enhancement features running in parallel to the wave flows. However, a notable difference in percentage wave reduction and velocity also occurred between the Rough tiles of different orientations. Surface roughness orientated perpendicular to the flow showed a greater impact in wave velocity reduction compared to tiles orientated with the flow. To date, this has not been tested or considered particularly for the presence of new hard enhancement features, and therefore this study provides valuable physical information regarding the engineering function of enhancement features as well as their role in promoting biodiversity. Overall, Chapter 5 demonstrates that 1) the presence of natural species should be considered and encouraged within coastal protection schemes where possible to aid in providing more environmentally friendly engineering and promoting biodiversity; and 2) increased complexity and the use of enhancement features not only increases biodiversity on hard artificial structures, but also fulfils an engineering function through wave velocity reducing properties. The increase in biodiversity can also further promote wave reduction due to the presence of colonising species, therefore providing a multifunctional role.

The overall results of my thesis provide valuable evidence for the types of multifunctional structures, novel materials and practical benefits enhancement features contribute to engineering and biodiversity. Enhancement features are a successful method of increasing species diversity and abundance on artificial shores (Chapman and Blockley, 2009; Martins et al., 2010; Browne and Chapman, 2011; Firth et al., 2012; Perkol-Finkel and Sella, 2015; Ido and Shimrit, 2015; Evans et al., 2016), and are particularly useful on previously constructed CDS. However, new CDS designs should consider the role of the whole structure and improving the overall environmental footprint of CDS. Porous CDS should be given priority where possible as they provide valuable engineering functions, reduce environmental footprints and contribute sheltered cryptic habitats. The placement of CDS in higher intensity environments is inevitable due to their engineering function, and the use of enhancement features is currently only considered for external surfaces. Priority must be on the increase in sheltered areas, followed by improved complexity in those areas. Furthermore, while the benefits of enhancement features have focused on increased biodiversity, it is clear that the increased complexity provides improved multi-functionality.

6.1.3 Recommendations and Considerations

From my research (and previous studies) there are a number of design considerations that (where possible) should be included in coastal engineering designs to promote biodiversity and sustainability. Practitioners and policy makers should encourage engineers to use environmentally-sensitive methods within all engineering designs. However, knowledge and evidence must be shared and connected between researchers, industry and policy makers in order to successfully implement changes to legislation and within engineering work. Independent non-profit bodies such as the Construction Industry Research and Information Association (CIRIA) often act as ‘knowledge brokers’ (Naylor et al., 2012) by interpreting and bridging the gaps between researchers and practitioners, and providing independent guidance from policies and industry standards. However, as these are independent guidance, they are not mandatory requirements for industry practitioners to follow, and therefore do not carry the sufficient weight to encourage more environmentally friendly practices, and too often the price of designs is a top priority over the environmental benefits. Policy legislation (as outlined in Chapter 1 Section 1.4) and industry guidance such as the Shoreline Management Plans (DEFRA, 2006), should have more defined guidance and criteria related to artificial defence structures and the use of enhancement features. At this stage in the evolution of ecological engineering, there is a lack of evidence and long-term quantitative monitoring, and few published studies that have carried out enhancement monitoring past 24 months (Chapman and Blockley, 2009; Martins et al., 2010; Browne and Chapman, 2011; Chapman and Underwood, 2011; Firth et al., 2014; Ido and Shimrit, 2015; Perkol-Finkel and Sella, 2015). Therefore, it is difficult to provide quantified recommendations at this stage. Here I will address what I consider the key best practice lessons learnt from my research which engineers should consider for coastal defence schemes.

Structural porosity: One of the key findings from this research was the multifunctional role provided by porous CDS. Where possible, coastal engineers should use porous structural designs, as opposed to solid structures, in coastal defence schemes. Porous structures provide a multifunctional role in engineering protection and habitat provisions through their sheltered, internal areas. Porous structures also provide additional internal complexities through unit connections which support species such as mussels (Moreira et al., 2007). As yet, it is unknown what the optimum level of structural porosity is for enhancing biodiversity in artificial areas, however, this would likely vary depending on the local area conditions and be limited to the engineering requirements. From this research, any porosity as opposed to solid structures such as sea walls, are recommended.

Enhancement features: Previous studies discussed throughout this thesis have demonstrated increases in biodiversity using ecologically-sensitive methods to enhance structural complexities. Water retaining features should be included to structural surfaces through simulated rock-pools (Evans et al., 2016) or wall pools (Chapman and Blockley, 2009; Browne and Chapman, 2011). Structural surfaces should also incorporate grooves, pits, crevices and roughened surfaces (Martins et al., 2010; Firth et al., 2013b). Many traditional coastal engineering materials provide “complex” features through the manufacturing and quarrying processes, such as grooves from blasting, holes from lifting, and uneven surfaces. Engineers should endeavour to utilise units with these features and place/ orientate these units within structures to maximise the biological benefits. This can be through placement within the structure so features are located internally (if possible), positioned in the lower intertidal zone, or horizontally facing to retain water. Placement of enhancement features has also been demonstrated through this research to add engineering benefits.

Ecosystem services: Where possible, coastal protection schemes should incorporate intertidal or subtidal species before or as part of a hard defence structure. Presence of species before hard defence structures could help to reduce wave energy loading onto the structure, thus potentially reducing the size of structures required (reducing environmental footprint), increase the lifespan and reduce maintenance requirements through lower environmental pressures. Species with large surface areas and high flexibility should be prioritised. However, careful consideration of suitable and native species should be made and researched to ensure maximum benefit and survival.

Material: Material choice is fundamental in coastal engineering. Hard materials, such as granite, provide robust protection against strong environmental conditions. However, they also hinder the settlement of coastal species. Materials such as limestone which have a softer composition and endure surface erosion provide more beneficial substrates and complex textures for coastal species (Borsje et al., 2011; Coombes et al., 2011). Engineers should avoid hard materials, and use those which meet maritime engineering strength standards but with softer surface textures. A mixture of materials in the same structure could also provide structural and biological benefits. pH of a material is also fundamental when encouraging biological settlement, and engineers should try to use pH neutral materials (Perkol-Finkel and Sella, 2011). In addition, the use of mollusc shells as a material aggregate within concrete has been shown to increase species settlement, particularly of other mollusc species (Risinger, 2012). This should be considered when designing structures for placement in an area where mollusc species are present.

Working alongside industry sponsors Arup, I aim to produce a formal industrial guidance document which will advise engineers and practitioners how to achieve best practice in coastal engineering. This will be followed by a cost-benefit analysis of work and methods to date, and will be used to advise clients.

6.1.4 Future Work

Albert Einstein famously said "*If we knew what it was we were doing, it would not be called research, would it?*" As with all research, there is no success without generating new questions and areas for improvements, and in light of my research there are a number of highlighted areas and recommendations for future work.

One of the key issues identified in Chapter 2 was the number of replicates and comparative sites. Future work should firstly aim to increase the number of sampling sites. This could improve the statistical power of the studies, where logistical constraints played a large factor in the limitations of the experimental designs. In addition, a revision of the data collection methods is also recommended in Chapter 2 to optimise surveying, collection and comparative analysis. Biological sampling using the SACFORN scale was deemed suitable during the experimental design stage, however in hindsight, percentage cover would provide more robust data to analyse. Equal numbers of biological and topographic sampling at the same sampling points would also enable stronger statistical analysis to determine any relationship between the two groups of samples at each spatial scale. Alternative topographic sampling methods could also allow comparison between different scale surface rugosity values. In this study I used the tape and chain method, more often used in coral reef assessments, it is well established but quite crude. A more technical and higher accuracy method could be carried out using digital processes such as LIDAR, high resolution drone mapping and 3D scanning. This would enable digital mapping of the structures and more intricate numerical analysis of the rugosity to carried out. Moreover, equal numbers of samples between spatial scales would enable better comparison of biological and topographic data, and enable full analysis of the results. In this study, the difference between sampling numbers hindered the ability to conclude whether spatial scale actually played a significant role in structuring biological communities. Furthermore, quantitative recordings of micro-habitat abundance and type would provide supporting information on habitat preference at different scale levels to Chapters 2 and 3. Routine data collection could also help to improve accuracy and provide informative results. Regular and seasonal sampling would allow for comparative studies over time and to consider seasonal variations in species. Whilst this study has provided useful insight into the characterising species in natural and artificial communities, it has also opened up new questions and areas for future studies. Future work should aim to fully determine whether spatial scale does play a significant role within structuring community data, and quantitative assessments of micro-habitats across different sites (natural and artificial) is recommended. These are two knowledge gaps highlighted through this study that could provide evidence supporting further development and implementation of ecological enhancements and designing more environmentally friendly defence schemes.

In Chapter 3, only one structure was sampled in this study due to the opportunity available. While this provided valuable insight into a previously unconsidered habitat area (internal cavities), further studies on other porous CDS would provide additional evidence to support the outcomes and strengthen the results found from this study. Furthermore, I would recommend either a similar decommissioning process in order to observe and record an established community on a structure, or the construction and observation of a new structure. The latter would allow full experimental design control, data collection from time zero and recordings of all activities experienced by the structure. However, limitations to recording the communities over time would be access to the internal compartments of the structure, potential damage to the structure, and extensive time required to allow communities to fully establish. Whilst this study showed that porous CDS provide a previously unconsidered area which can function as a potentially suitable habitat for coastal species, it has also opened up new areas for future studies. A key knowledge gap identified from this study is the level of porosity required to provide suitable habitats for biological species on artificial structures. Whilst there may be some limitations to the porosity possible when designing a porous CDS due to its engineering role, a better understanding as to what level/ range of porosity is effective at providing artificial habitat spaces would be valuable for advising engineers designing new schemes.

In Chapter 4, issues stemmed from the fragility of the small-scale designs, fixing methods and design specification limits related to coastal environments. Future studies should consider using larger scale prototypes to test the design specification limits fully in order to determine the true limitations for use in coastal engineering. In addition, alternative fixing methods should be trialled to explore effective methods of increasing the tile *survival* and enabling the incorporation of 3D printing hard structural "accessory" units. Other fixing methods were considered as part of this study, but financial, time and logistical limitations resulted in the decision to use a screw fixing methods. Future use of 3D printed accessory units should also consider the placement of the units within the structure. My results show susceptibility of the material to erosion due to wave exposure. However, placement of the units in more sheltered areas will reduce erosion rates and encourage colonisation of species as demonstrated in Chapter 3. As this experiment was largely unsuccessful due to the loss of the tile units, and I was unable to address whether 3D printed material is a suitable material for encouraging biological communities, I would recommend future work repeats the experiment again but with a revised experimental design. Firstly, a more suitable location should be considered with less coarse gravel material and lower wave exposure. Secondly, refinement to the designs should be made, avoiding designs with fine features that are more fragile and susceptible to breaking. In addition, an alternative fixing method is advised. Marine cement would be the most ideal, or fixing the tile units using a metal bracket frame to avoid adding weakness to the units. Thirdly, comparisons of the biological settlement should be compared to other artificial and natural communities. Time-zero experiments across the tile units and a select number of natural and artificial sites of varying material but with similar environmental conditions would provide valuable insight into the effectiveness of 3D printing material, and comparisons to other materials which is essential. Comparisons will also enable leverage in promoting this technique to engineers, other practitioners and policy makers if the experiments show positive results.

Lastly, following the studies in Chapter 5, which provided a standard comparative experiment with no scaling, larger scale physical models are recommended to minimise scaling issues often caused by the lack of similarity in wave reflection, transmission, energy frictional dissipation and wave breaking (among others) between models and the real world designs (Hughes, 1993; Chadwick and Morfett, 1998). This would help to determine full capabilities of the use of species in soft engineering. Suggestions and discussions with *Deltares*, a Dutch hydraulics research company, recommended using the Danish Hydralab to carry out full-scale physical model tests demonstrated by Möller et al. (2014). In addition, computer fluid dynamic (CFD) models could provide further evidence as they enable more comparative variations and changes in parameters quickly compared to physical models. They also remove scaling issues by running numerical models based on full-scale designs. However, CFD modelling requires a large amount of assumptions and detailed designs in order to provide accurate representation of real life scenarios, therefore it would be (initially) difficult and time intensive for carrying out a study such as this which compares several different species. Lastly, careful consideration and an increased knowledge of the interactions between ecosystem engineering species and wave attenuation is important to: 1) understand better their role in coastal protection (Ysebaert et al., 2011b), and 2) to understand the biological requirements and limitations of different species in order to successfully select suitable species for areas where they do not currently exist. Whilst this study provided valuable insight into the different physical properties of species which are more effective at reducing wave energy, and the multifunctional role that enhancement feature can play on CDS, there are new questions that have been exposed and should be addressed in future. The key knowledge gaps identified through this study that should be addressed in future work are the potential of different species and enhancement features at a full scale level with real wave conditions. It is recommended that future studies aim to develop knowledge of different species and provide comparative guides of the different physical properties, their effectiveness at reducing wave energy (and to what extent i.e. wave height), and any biological limitations, that can be used by engineers.

6.2 Research Impacts

A fundamental part of undertaking an EngD as opposed to a traditional PhD, is the industrial partnership and applicability of the research to a wider audience. Ensuring that my research provided industrial relevance and benefits was essential, in addition to the contribution of new knowledge and techniques through novel studies. Through funding and collaboration with industry sponsors, *Ove Arup & Partners* and *Fugro*

EMU, my research is being used in a number of maritime schemes and industrial documents. Through this research I was recruited to work within the Arup research team and develop the marine environmental business bridging maritime engineering and environmental consulting. To date, successful engagement with a number of different project clients has resulted in modifications to traditionally proposed maritime designs with more environmentally sensitive techniques developed both through my research and previously considered methods. Working with Thames Tideway (and contractors), I have produced new design principles for three site developments using this research which are being incorporated into the new designs. Porous rock armour toe scour units and wall "accessory" units are being designed and placed on existing river wall structures to add habitat complexity, and mimic species using geotextiles are being considered for incorporation to reduce wave flow and erosion around new outfall structures.

In addition, Arup have been successful in two large bid schemes, Portsmouth City Council Coastal Flood & Erosion Framework and Aberdeen Harbour, and are in the process of a further two bids for Guernsey Harbour and Manila Bay, Philippines. As a result of my knowledge development, I was requested to provide technical contributions to the bids, including relevant aspects of this research such as suggesting alternative ecologically sensitive designs in the bid application, and been appointed as a core team member in the winning bids. Other ongoing projects which have requested ecological enhancement features to promote biodiversity and add aesthetic value to their artificial structural designs, are: London Garden Bridge, UK; Swansea Bay Tidal Lagoon, UK; Jurong and East Coast Parkway Coastal Development, Singapore; and an eco-tourism development in Honduras. Engagement and consultation with project leaders is ongoing and these projects will require modifications to current design methods and incorporation of new habitat enhancing features. Furthermore, their high profile status has enabled us (Arup) to propose new novel designs and research ideas to trial new experiments in different locations, and gain practical evidence to develop the field of ecological engineering.

The business and environmental values of my research and the new knowledge to projects has resulted in the publication of my research in Arup's UKMEA Sustainability Report (Arup, 2016). This report is an external facing and internationally distributed document that promotes Arup's most innovative and sustainable engineering methods being used in their projects. In addition, my research is being included in the REFINET (REthinking Future Infrastructure NETworks) project, a Coordination Support Action (CSA) funded by the European Commission that aims to integrate transport infrastructure sectors and create a shared vision of "...specified, designed, built or renovated, and maintained multi-modal European transport infrastructure network of the future". As part of this project my research will be included as a case study for generating sustainability guidelines and design principles for relevant transport infrastructure across

Europe. In 2015, I was invited to present my research at London's International Shipping Week and assist in running a workshop on sustainable infrastructure. Finally, my research was also featured in the Underwater Technology (UT3) Magazine (SUT, 2012). Such industrial publications are essential to support, demonstrate and bridge the gaps between research and industrial applications. These industrial outcomes are in addition to the academic conferences and workshops which have lead to discussions for future potential collaborations and project opportunities.

Despite the growing interest and support from industrial partners, practitioners and policy makers, who are seeking opportunities to become industry leaders and to promote their '*environmental*' and sustainable approaches to engineering schemes, there still remains some hesitancy and questions regarding practicality and logistical constraints when discussing design enhancements, such as:

1. How long do enhancement features or natural methods last?
2. How long do they take to establish good species colonisation?
3. What are the cost benefits to each method?
4. How much enhancement is "enough"?

To continue developing industry and research links, these are fundamental questions that must be addressed. Successfully reducing the environmental impacts from CDS can only be achieved through collaborations and shared knowledge, therefore it is vital that researchers engage more with industries, policy-makers and environmental practitioners to bridge knowledge gaps and incorporate research within schemes to generate more robust and practical evidence in this areas. Industrially focused guide books such as CIRIA (2015, 2016); Arup (2016) are important documents to bridge the gaps, promote best practice and innovative solutions, and ensure relevant industries are kept up-to-date with academic progress.

6.3 Steps to practical implementation of multifunctional coastal defence structures

The development of ecological engineering in the marine environment has provided a series of valuable 'proof-of-concept' studies demonstrating the environmental benefits of enhancement features on artificial CDS (Chapman, 2003; Moschella et al., 2005; Chapman and Blockley, 2009; Martins et al., 2010; Browne and Chapman, 2011; Chapman and Underwood, 2011; Firth et al., 2012; Carvalho et al., 2013; Aguilera et al., 2014;

Firth et al., 2014; Perkol-Finkel and Sella, 2015; Ido and Shimrit, 2015; Evans et al., 2016; Firth et al., 2016). Although there are clear positives for incorporating these features into CDS designs, there is still limited awareness among engineers and practitioners, which must be overcome in order to practically implement such modifications. This also extends to the use of soft engineering methods and in particular ecosystem engineering species. In general, this is due to a lack of evidence and long-term quantitative monitoring, particularly in intertidal enhancements where there are higher environmental pressures (but see Harris, 2003; Hartig et al., 2011; Jackson and Tomlinson, 2012; Mendonça et al., 2012; Scyphers et al., 2015). The importance of long-term monitoring in marine studies is well acknowledged (Hawkins et al., 2013a,b), and particularly in application to ecological engineering it enables us to develop a full understanding of how enhancement features perform over long time periods to provide true best practice guidance. However, at this stage in the evolution of ecological engineering, there are few published studies that have carried out enhancement monitoring past 24 months (Chapman and Blockley, 2009; Martins et al., 2010; Browne and Chapman, 2011; Chapman and Underwood, 2011; Firth et al., 2014; Ido and Shimrit, 2015; Perkol-Finkel and Sella, 2015). Although results and publication at an early stage provide indicative evidence, there are still a number of key questions that can only be answered through longer term monitoring: e.g. the development of coastal biological communities; the impacts on longevity or increased lifespan of CDS through the use of ecological techniques; or facilitation of coastal species through changing environmental conditions. More so, a lack of long-term evidence limits the ability to provide accurate cost-benefit analysis on different techniques, and advising on the '*optimum*' level of enhancements within structures, a factor which has not yet been considered. Working more closely with industry partners who regularly monitor and record coastal data, for example the Channel Coastal Observatory, local ecology groups and council coastal engineers, more routine sampling of defence structures could be carried out. Through repetitive sampling more data relating to artificial biological communities, anthropogenic activities and the development of enhanced structural features, can be recorded and shared amongst relevant stakeholder. Groups such as the Eastern Solent Coastal Partnership are actively encouraging local councils and volunteers to help monitor coastal biodiversity and damage to CDS.

In addition, there is still little consideration regarding the indirect impacts of CDS such as anthropogenic disturbance. Anthropogenic disturbance covers a range of abiotic pressures from maintenance, trampling, and transport infrastructure (e.g. coastal railways, ports and harbours), and is a critical factor affecting the distribution, abundance and composition of colonising species on artificial structures (Airoldi and Bulleri, 2011). It is well understood that disturbance events open up further prospects to opportunistic and invasive species and can subsequently decrease native dominant species (Dayton, 1971; Sousa, 1979; Hutchinson and Williams, 2003; Tsinker, 2004; Airoldi and Bulleri, 2011). However, to date there has been little research looking at the full extent of anthropogenic disturbance and methods for minimising it (Airoldi and Bulleri, 2011). This research touched briefly on the maintenance impacts experienced during engineering works (Chapter 3), and noted extensive damage and loss of colonised surface areas. Activities related to maintenance works are inevitable, however, with effective research and management methods, the impact of these engineering works can be reduced providing a number of social, economic and environmental benefits. This is a key area which should be investigated in future research and an important area to be considered in future marine policies and planning.

6.4 Concluding comments

As the demand for increased coastal protection grows due to environmental changes, the environmental implications must be considered. Ecologically sensitive coastal defence structures (CDS) are essential to promote biodiversity and sustain our coastal resources. Research on marine ecological engineering methods has shown successful techniques for promoting biodiversity on artificial structures. This research sets out to develop methods of designing multifunctional CDS that provide environmental and engineering roles. The overall results of this thesis provide valuable evidence for designing multifunctional structures, and have shown the benefits of porous CDS and the cryptic internal habitats they provide; and the use of enhancement features for improving engineering functions and biodiversity. New CDS designs should consider the role of the whole structure and improving the overall sustainability of CDS. Porous CDS should be given priority where possible as they provide valuable engineering functions, reduce environmental footprints and contribute sheltered cryptic habitats. The placement of CDS in higher intensity environments is inevitable due to their engineering function, and the use of enhancement features are currently only considered for external surfaces. Priority must be on the increase in sheltered areas, followed by improved complexity in those areas. Furthermore, whilst the benefits of enhancement features has been on increased biodiversity, it is clear that the increased complexity provides improved multi-functionality.

Finally, the proliferation of hard coastal protection can only be minimised to a certain extent without diminishing the engineering functionality. More consideration needs to be given to using soft engineering approaches instead of, or alongside, hard engineering structures. There is a growing evidence for better planning and proactive approaches to implementing research outputs within industry scheme, however, until there is better collaboration and communication between academics, engineers, policy makers and practitioners, there will continue to be large gaps and loss of more coastal areas.

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Appendices

Appendix A

Review of ecological enhancement studies

Table 1: Research and commercial products for coastal enhancement and protection from a micro-scale testing materials surface texture through to macro-scale at structural design, and intertidal and sub-tidal techniques. Techniques in italics indicate work covered in this thesis.

Engineering	Study Type	Location	Result	Reference
Materials				
Texture and structure of concrete constructions	Academic	Ijmuiden, The Netherlands	Fine or coarse surface were colonised more rapidly by small green algae than smoother surface. Cups and holes favoured the initial colonisation by larger green algae. Slabs in the mid and low tidal zone were rapidly overgrown by barnacles. Mussels and periwinkles were found in the sections with grooves, and holes and cup.	(Borsje et al., 2011)
Surface texture of construction materials	Academic	Cornwall, UK	Significant differences between microbial growth features and micro-erosion between limestone, granite and marine concrete. Limestone displayed the largest amount of euendolithic borehole erosion and inorganic weathering, followed by concrete which mostly consisted of biochemical crusts. Granite displayed thin epilithic films.	Coombes et al. (2011)

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Engineering	Study Type	Location	Result	Reference
Oyster shell breakwaters - various designs	Academic	Louisiana, USA	Oyster spat fall exceeded 10,000 spat/m^2, with shell heights measuring 50 cm after six months of growth and exceeding 500 per m^2 on the artificial breakwater reefs. Biologically dominated concrete structures showed a significant increase in flexural strength over time compared to traditional aggregates.	Risinger (2012)
Ecologically active & textured concrete	Industry	Red Sea & Ashdod, Israel	Modified concrete compositions and surface textures can enhance the colonisation of marine fauna and flora. This in turn can improve water qualities, structural life span and stability, and reduce hydrodynamic forces.	(Perkol-Finkel and Sella, 2011)
Micro-habitats				
Addition of different size pits to sea wall	Academic	West Sussex, UK	Increased diversity and complexity, and larger abundance of barnacles in rough surfaces and small crevices than smooth surfaces.	Moschella et al. (2005)
Crevice spaces between rock units	Academic	Kirribilli, Australia	Increased densities of chitons in crevices than exposed surfaces (over 50% more).	Moreira et al. (2007)

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Engineering	Study Type	Location	Result	Reference
Added cavities to sea wall to create water retaining shaded recesses	Academic	McMahns Point, Australia	Larger number of species in artificial pools than nearby natural with increased diversity of sessile and mobile organisms, and increased foliose algae.	Chapman and Blockley (2009)
Drilled pits of various sizes and densities	Academic	Azores, Portugal	Up to 10 times as many mobile limpets found in pits.	Martins et al. (2010)
Mimic rock pools using flower pots attached to sea walls	Academic	Sydney, Australia	Presence of cavities and artificial rock-pools on sea walls increased numbers of species by 110% within months. Mobile fauna, in particular are most affected by replacing natural shores with walls.	Browne and Chapman (2011)
Sloping wall of small blocks and boulders to sea wall	Academic	White Bay/ Quakers Hat Bay, Australia	No increase in biodiversity from either method, but decrease in sessile and mobile species on horizontal than vertical surfaces.	Chapman and Underwood (2011)
Addition of holes and grooves to sea walls	Academic	Farm Cove, Australia	Increased number of limpets and chitons in grooves than holes and normal surface.	Chapman and Underwood (2011)
Added crevice spaces between rock units	Academic	Kirribilli, Australia	Increased species richness of algae and sessile invertebrates in crevices than exposed surfaces.	Dugan et al. (2011)

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Engineering	Study Type	Location	Result	Reference
Addition of grooves, pits and crevices to structures	Academic	Plymouth/ Shaldon, UK	Increased species richness with majority of species unique to pits, and barnacles only found in recess areas.	Firth et al. (2013b)
Drilled cored rock pools of two different depths	Academic	Tywyn, UK	Greater species richness in drilled-core rock pools than adjacent granite rock surfaces on the breakwater, and similar species richness to natural rock pools on nearby rocky shores. More species found in shallow drilled-core pools (30% more) than deeper pools. Different community composition between artificial and natural pools.	Evans et al. (2016)
Mimic rock pools using 'Vertipool' designs attached to sea walls	Academic/ Industry	Isle of Wight, UK	No results - experiments are ongoing.	Personal contacts - Ecclestone George ®, Artecology ®, Bournemouth University

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Engineering	Study Type	Location	Result	Reference
Mimic rock pools using inverted tyres attached to sea walls	Academic/ Industry	Isle of Wight, UK	No results - experiments are ongoing.	Personal contacts - Ecclestone George ®, Artecology ®, Bournemouth University
Habitat units				
Pre-cast BIOBLOCK unit incorporated into rip-rap structures with different habitats	Academic	Colwyn Bay, UK	The BIOBLOCK consistently supported greater species richness than the adjacent boulders, particularly after 13 months. Deep pools supported the highest number of species, followed by shallow pools, deep pits and then shallow pits supporting the lowest species richness.	Firth et al. (2013b)

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Engineering	Study Type	Location	Result	Reference
Pre-cast habitat rock pool units incorporated into rip-rap structures	Industry	Haifa, Israel	The abundance, richness and diversity of species were higher on and around the ECOncrete ® designed armour units compared to standard units, in particular ecosystem engineering species such as oysters, serpulid worms, bryozoans and coralline algae were more abundant. The presence of invasive species was also lower.	Ido and Shimrit (2015)
Textured ECOncrete ® units - Pier piles and tide-pools	Industry	New York, USA	Applying innovative technology of ecological concrete encasement and water retaining tide-pools, valuable habitats are created and an increase in species richness and abundance.	(Perkol-Finkel and Sella, 2015)

Appendix B

Review of ecological engineering studies

Table 2: Examples of previous research carried out using coastal ecosystem engineers to improve environmental conditions.

Species	Type of Study	Benefits	Considerations	Reference
Plants				
Bolboschoenus maritimus	Field - Elbe estuary, Germany	Reduce flow velocity under normal conditions. Plants on the edge of the marsh with larger diameter reduce wave velocity more than 15%. Positive correlation between plant thickness and cross-shore currents.	Wave energy stress can damage plant structure and characteristics.	Carus et al. (2016)
Spartina alterniflora (Salt marsh)	Laboratory	Wave attenuation appeared to be most dependent on stem density and the ratio of stem length to water depth. Wave attention increased slightly with wave height while no clear trend with respect to wave period was seen. Wave energy loss occurred at all frequencies of both spectral types, with dissipation increasing with frequency above the spectral peak.	An empirical relationship defining the bulk drag coefficient for *S. alterniflora* as a function of the stem. Reynolds number is found to serve as a first estimate for engineering applications.	Anderson and Smith (2014)

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Species	Type of Study	Benefits	Considerations	Reference
Spartina alterniflora (smooth cordgrass)	Laboratory	Comparing with experiments without vegetation, it is estimated that up to 60% of observed wave reduction is attributed to vegetation.	Although waves progressively flatten and break vegetation stems and thereby reduce dissipation, the marsh substrate remains stable and resistant to surface erosion under all conditions.	Möller et al. (2014)

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Species	Type of Study	Benefits	Considerations	Reference
Phragmites australia (Reeds)	Field - Baltic Sea	Wave height attenuation recorded a maximum of 2.6% m $^{-1}$ and 11.8% m $^{-1}$ from open water into the reed vegetation. Wave attenuation through the emergent reed vegetation was lower in greater water depths, suggesting (1) a reduced influence of bed friction by small shoots/-roots and/or (2) drag reduction due to flexing of plants when the wave motion is impacting stems at a greater height above the bed. Wave dissipation increased with increasing incident wave height, suggesting that despite their ability to flex, reed stems may be rigid enough to cause increased drag under greater wave forcing.		Möller et al. (2011)

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Species	Type of Study	Benefits	Considerations	Reference
Salt marsh & macrophytes	Field - Yangtse Estuary, China	Vegetation can reduce wave heights up to 80% over a relatively short distance (<50 m). *Spartina alterniflora* is able to reduce hydrodynamic energy from waves to a larger extent than *Scirpus mariqueter*, and therefore has a larger ecosystem engineering capacity (2.5 higher on average). A higher standing biomass of S. alterniflora explained its higher wave attenuation at low water depths. Being much taller compared to *S. mariqueter, S. alterniflora* also attenuated waves more with increasing water depth.	Knowledge about the engineering properties of salt marsh species is important to better understand wave attenuation by tidal wetlands and their possible role in coastal protection.	Ysebaert et al. (2011b)

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Species	Type of Study	Benefits	Considerations	Reference
Spartina spp. & *Salicornia* spp. (Salt marsh)	Field - Dengie Peninsula, UK	Vegetation density/type did not have a significant direct effect on wave attenuation but modified the process of wave transformation under different hydrodynamic conditions. *Spartina* spp. - relative incident wave height identified as a statistically significant dominant positive control on wave attenuation up to a threshold value (0.55), beyond which wave attenuation showed no significant further increase. *Salicornia* spp. - no significant relationship existed between wave attenuation and relative wave height.	Seasonally (between September and December) significant temporal increase/decrease in vegetation density occurred in one of the *Spartina* canopies and in the *Salicornia* canopy, respectively, and led to an expected (but not statistically significant) increase/decrease in wave attenuation.	Möller (2006)

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Species	Type of Study	Benefits	Considerations	Reference
Halodule wrightii, *Syringodium filiforme*, *Thalassia testudinum* and *Zostera marina* (sea grass)	Laboratory	Wave energy reduction per meter of sea grass bed equalled 40% when the length of these sea grasses was similar to the water depth.	Sea grass wave energy reduction is similar to salt marshes when water depth is same as plant size. Broader, shallow meadows of sea grasses create substantial wave energy reduction.	Fonseca and Cahalan (1992)
Bivalves				
Crassostrea virginica (Eastern oyster) & *Spartina alterniflora* (smooth cordgrass)	Laboratory	Treatments of 1-year old showed more energy attenuation than newly-deployed treatments, and a combination of 1-year old *S. alterniflora* and *C. virginica* was the most effective treatment reducing 67% of the wave energy.	Little information is available on the success of ecosystem shoreline stabilisation	Manis et al. (2015)

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Species	Type of Study	Benefits	Considerations	Reference
Mussels	Field - Wadden Sea, Netherlands	Large increase in measured bed shear stress over the mussel bed compared with the uncovered parts of the intertidal flat. This is caused by the large roughness of the mussels.	Mussel settling phase to the substrate is critical in mussel bed development, and the initial attachment period is the strongest.	Donker et al. (2012)
Mytilus edulis (mussels) & *Crassostrea gigas* (Oysters)	Laboratory	For the same physical forcing, natural oyster beds are more effective in wave attenuation (50% reduction in wave height) compared to natural mussel beds (30% reduction in wave height).	Both reef building species can contribute to (1) stabilising the bed of intertidal flats in front of dykes, (2) enhance biodiversity by providing shelter and nesting area for fish and crustacean species (3) clarify the water by removing particulates from the water column	Borsje et al. (2011)

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Species	Type of Study	Benefits	Considerations	Reference
Oysters	Field - Alabama, USA	Oyster breakwater treatments showed mitigation of shoreline retreat by more than 40% at one site. Adult oyster densities reached more than eighty oysters m^{-2} at one site.	Overall vegetation retreat and erosion rates were high across all treatments and sites.	Scyphers et al. (2011)
Crassostrea virginica (Eastern oyster)	Field - Louisiana, USA	Shoreline retreat was reduced in cultched low-energy shorelines compared to the control low-energy shorelines, but was not significantly different in high-energy environments.	Oyster spat grew at a rate of 0.05 mm/day.	Piazza et al. (2005)
Oysters	Field - California, USA	Significant differences in sediment erosion and accretion were observed, particularly after a major storm, between areas with and without oysters present and not present. Marsh edge in the cultched treatment averaged 2.9 cm of accretion and the non-cultched an average loss of 1.3 cm.	Site orientation and anthropogenic disturbance also influenced sediment erosion and accretion rates.	Meyer et al. (1997)
Coral Reefs				

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Species	Type of Study	Benefits	Considerations	Reference
Coral reefs	Numerical model	Refraction analysis shows that different reef shapes produce characteristic patterns of wave convergence on reef surfaces. Location and stability of wave convergence is largely controlled by the shape of platforms, which further controls the distribution of wave energy across platform surfaces.	Wave propagation defines sediment transport vectors and control the transport and deposition of different sized material. Platforms such as elliptical and circular reefs, are more likely to retain sediment on reef surfaces by promoting marked wave convergence behaviour.	Mandlier and Kench (2012)
Mimics				
Spartina alterniflora (Salt marsh)	Laboratory	Using polyolefin tubing as *S. alterniflora* mimics, recorded wave attenuation appeared to be most dependent on stem density and the ratio of stem length to water depth. Wave attention increased slightly with wave height while no clear trend with respect to wave period was seen.	An empirical relationship defining the bulk drag coefficient for *S. alterniflora* as a function of the stem. Reynolds number is found to serve as a first estimate.	Anderson and Smith (2014)

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Species	Type of Study	Benefits	Considerations	Reference
Mangrove trees	Laboratory	Using artificial mangrove models, a reduction in wave height in areas with mangroves was observed to be two times larger compared to that in bare areas. Tandem and staggered pattern arrangements of trees showed <10% difference and *Rhizophora* forest of 80 m in width reduce wave height by 80%.	Quantitative effects of mangrove vegetation characteristics are poorly understood.	Hashim and Catherine (2013)
Posidonia oceanica (Sea grass)	Laboratory	Artificial *P. oceanica* meadow were configured in various patterns and tested two stem density patterns and height. Wave orbital velocities showed significant attenuation inside the meadow and above the flume bed. Energy transfer near the meadow edge, showed spectral wave velocities from the longer to the shorter wave period components.		Koftis et al. (2013)

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Species	Type of Study	Benefits	Considerations	Reference
Zostera noltii (Sea grass)	Laboratory	There is positive correlated between wave attenuation and blade stiffness, and for a given wave in shallow water. Attenuation is dependent on a combination of shoot density and leaf length (leaf area index). Tidal currents can strongly reduce wave-attenuating capacity of sea grass mimics, and was most pronounced at high shoot densities.	Structural overestimation of wave attenuation for tidal environments, emphasising that tidal currents need to be taken into account in future studies on wave attenuation by vegetation.	Paul et al. (2012)
Wetland vegetation	Laboratory	Emergent conditions resulted in a higher wave attenuation compared to near-emergent conditions most likely because the plant stems occupy the entire depth of the water column, whereas near-emergent conditions stems do not impede the top portion of the water column.	Wave attenuation from rigid and flexible vegetation elements were very similar in the laboratory experiments and yielded the same friction factors.	Augustin et al. (2009)

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Species	Type of Study	Benefits	Considerations	Reference
Coral reefs	Laboratory	A submerged uniform array of rods mimicking coral reefs showed to decrease the flow velocity due to increase in the bottom drag coefficient, which is a strong function of the canopy porosity.	The exit flow velocity from the porosity gaps is significantly higher compared to the surroundings, thus leading to jetting flow. Removal of natural barriers may cause local flow intensification, leading to negative impacts on coastal areas.	Fernando et al. (2008)
Kelp	Numerical & laboratory	The expressions for the wave number and the exponential decay coefficient derived for arbitrary damping are shown to reduce to those based on linear wave theory and the conservation equation of energy if the damping is small.	Calibrated drag coefficients for these runs are found to vary in a wide range and appear to be affected by the kelp motion and viscous effects that are neglected in the analysis.	Kobayashi et al. (1993)

Appendix C

List of species recorded in Chapters 2 & 3

Table 3: Species and morphotaxa recorded in Chapters 2 & 3 from sites on the south coast and Isle of Wight, UK.

Species and morphotaxa	
Brown macroalgae	**Barnacles**
Algaozonia	*Semibalanus balanoides*
Ascophyllum nodosum	*Chthamalus* sp.
Dictyota dichotoma	
Fucus serratus	**Lichen**
Fucus spiralis	*Pyrenocollema halodytes*
Fucus vesiculosus	
Sargassum muticum	**Mobile crustaceans**
	Carcinus maenas
Green macroalgae	*Carcinus maenas* (juv)
Ulva lactuca	
Ulva prolifera	**Molluscs**
	Buccinum undatum
Red macroalgae	*Callistoma zizyphinum*
Chondrus crispus	*Gibbula umbilicalis*
Corallina officinalis	*Littorina obtusata*
Corallinaceae crust	*Mytilus edulis*
Erythroglossum laciniatum	*Nucella lapillus*
Hildenbrandia sp.	*Patella depressa*
Mastocarpus stellatus	*Patella ulyssiponensis*
Osmundea pinnatifida	*Patella vulgata*
Polysiphonia sp.	
Porphyra sp.	**Polychaetes**
	Eulalia viridis
Anemones	
Actinia equina	
Actinia fragacea	

Appendix D

D-Shape **Equipment technical data**

CAD-CAM procedure

D-Shape **Chemical technical data**

D-Shape printer

Measuring 6 m x 6 m with around 300 ink jet nozzles, the *D-Shape* equipment is suspended on a gantry on-top of four raised columns. The gantry runs along a frame where it provides two functions: 1) flattening of material, 2) depositing the binder onto the material. Each ink jet nozzle releases approximately 5 mm (in width) of binder, with around 20 mm between each nozzle. Each layer is produced in four steps, by shifting each step by 5 mm to ensure the design space is adequately covered with binder for solidification (refer to Jakupovic (2013)).

D-Shape Equipment technical data

Technical details of the D-Shape 3D printing technology are listed below (adapted from Jakupovic (2013)):

- **Height:** 3-12 m
- **Printing area:** 6 x 6 m
- **Number of nozzles:** 300
- **Nozzle reaction time:** 10-15 ms
- **Weight (without substrate feeder):** 1,300 kg
- **Total weight:** 5,000 kg
- **Voltage:** 380-220 V, 50 Hz
- **Energy consumption (without substrate feeder):** 2 kW
- **Total energy consumption:** 40 kW
- **Theoretical productivity:** 6 min/ layer
- **Pixel size:** 5 mm
- **Layer thickness:** 5 mm $pm0.5$ mm
- **Theoretical resolution:** 25 dpi
- **Effective resolution:** 4-6 pdi

CAD-CAM procedure

Full operating procedure from virtual design to material production is outlined as follows (from Jakupovic (2013)):

1. Creation of a three dimensional CAD file
2. Conversion of the CAD file to the .stl format
3. Opening of the file in software called *Monolite* (proprietary program), which operates the following:
   (a) Slicing of the three dimensional volume into 5 millimeter layers
   (b) Dividing each layer into dots representing the drops of liquid
4. Deposition of the substrate on the printing surface (external to the machine)
5. Spreading and flattening of a layer of substrate
6. Jetting of the binder on the layer in four steps
7. Elevation of the four columns by 5 mm and the return to step 4.
8. Curing of the material, once all the layers have been printed
9. Removal of the supporting powder bed (external to the machine, mostly manual)

D-Shape Chemical technical data

Technical *D-Shape* material composition information is as follows (from Shand (2007); Jakupovic (2013); Dinitech (2013a,b)):

- **Chemical nomenclature:** Magnesium Oxide
- **Chemical formula:** MgO
- **Physical state:** White powder
- **Molecular weight:** 40,3
- **Hydrosolubility (at 18°C):** 86
- **pH (water suspension 100 g/l at 18°C):** 10.5
- **Melting point:** 2.5-2.8 °C
- **Boiling point:** 3.6 °C
- **Specific weight:** 3.58

- **Magnesium Oxide:** 84 ± 2 %
- **Calcium Oxide:** 2.35 ± 0.85 %
- **Other Oxides:** 0.45 ± 0.15 %
- **Calcination loss:** 3.5 ± 1.5 %
- **Insoluble residue:** 9.5 ± 1.5 %
- **Sieve residue (54 micr.):** 35 %
- **Chemical nature:** Magnesium Chloride water solution
- **Chemical formula:** $MgCl_2$*$(H2O)_n$
- **$MgCl_2$*$6H_20$ content:** 64/65 %
- **Physical state:** liquid, lightly amber colored
- **Odor:** Odorless
- **pH:** 6±1
- **Density at 20C:** 1.288 1.342 Kg/dm^3
- **Magnesium Chloride $MgCl_2$:** 29.5 - 31 %
- **Magnesium Sulfate $MgSO_4$:** 3.5 4 %
- **Potassium chloride KCl:** 0.3 0.4 %
- **Sodium chloride NaCl:** 0.05 0.1 %
- **Melting point:** 714 °C
- **Boiling point:** 1,413 °C

Sorel Cement chemical reaction between the two compounds is as follows:

$$5MgO + 1MgCl_2 + 13H_2O \rightarrow 5Mg(OH)_2 \cdot MgCl_2 \cdot 8H_2O(5phase)$$

Appendix E

3D Print Tile Replicate Set-up

Table 4: Forty four 3D printed tile units deployed at Highcliffe, Christchurch Bay, UK over two groynes on the far westerly side of the site.

Groyne No.	Sample No.	Tile Design	Elevation	Orientation
G1	1	Rough	Low	Horizontal
G1	2	Wiggles	Mid	Horizontal
G1	3	Control	Low	Vertical
G1	4	Rough	Low	Vertical
G1	5	Control	Mid	Horizontal
G1	6	Square	Top	Horizontal
G1	7	Rough	Mid	Slant
G1	8	Circles	Mid	Vertical
G1	9	Square	Top	Horizontal
G1	10	Wiggles	Mid	Slant
G1	11	Control	Low	Horizontal
G1	12	Wiggles	Low	Horizontal
G1	13	Circles	Mid	Vertical
G1	14	Control	Low	Vertical
G1	15	Circles	Mid	Horizontal
G1	16	Wiggles	Mid	Horizontal
G1	17	Circles	Low	Horizontal
G1	18	Square	Mid	Horizontal
G1	19	Rough	Low	Vertical
G1	20	Square	Mid	Horizontal
G1	21	Control	Low	Vertical
G1	22	Rough	Low	Horizontal
G2	1	Rough	Low	Horizontal
G2	2	Rough	Mid	Vertical
G2	3	Control	Mid	Vertical
G2	4	Circles	Mid	Slant
G2	5	Wiggles	Mid	Slant
G2	6	Control	Mid	Vertical

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Groyne No.	Sample No.	Tile Design	Elevation	Orientation
G2	7	Control	Mid	Vertical
G2	8	Rough	Low	Slant
G2	9	Circles	Mid	Horizontal
G2	10	Wiggles	Mid	Slant
G2	11	Circles	Mid	Vertical
G2	12	Wiggles	Top	Horizontal
G2	13	Square	Low	Horizontal
G2	14	Square	Mid	Slant
G2	15	Wiggles	Low	Slant
G2	16	Circles	Mid	Slant
G2	17	Control	Top	Horizontal
G2	18	Square	Top	Horizontal
G2	19	Control	Top	Horizontal
G2	20	Rough	Mid	Horizontal
G2	21	Rough	Mid	Horizontal
G2	22	Square	Mid	Horizontal

Appendix F

Publications

Hidden biodiversity in cryptic habitats provided by porous coastal defence structures.

Sherrard et al. (2016)

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Maintenance Activity and the hidden habitats in coastal defence structures: new steps towards effective ecological management

Sherrard et al. (2015)

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Hidden biodiversity in cryptic habitats provided by porous coastal defence structures

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A B S T R A C T

In response to flood risk from rising and stormier seas, increasing amounts of natural coastline worldwide are being replaced by a proliferation of coastal defence structures. While the primary role of defence structures is protecting the coastline, consideration should be given to the biological coastal communities they support. Artificial structures are currently seen as poor habitats for marine organisms. They are constructed in harsh coastal environments, lack structural complexity, and are subjected to episodic disturbance from maintenance, reducing their suitability as habitats for coastal species. Recent work has focused on mitigating the impacts of coastal defence structures, through secondary routes such as enhancing biodiversity by encouraging colonisation of marine biota. Research thus far has focused on enhancements to improve structural complexity on the external surfaces of coastal defences. Many structures are porous with internal compartments. To date no work has been undertaken on the habitat provided by the internal surfaces of the blocks used in building structures.

We investigated the role of porous coastal defence structures in habitat provision. Taking advantage of a groyne reduction from 45 m to 20 m length, we surveyed the internal environment of the structure. We also considered the impacts of maintenance activity on coastal assemblages. Our work shows that the internal environment of artificial structures provides functional habitat space supporting higher species richness and diversity than external surfaces. The more benign environment of internal surfaces protects from desiccation stress and is probably less scoured by mobile sediments, and as such is of unrealised importance to coastal assemblages. External surfaces are also subject to high levels of disturbance from maintenance activities, further limiting the potential ecological contribution this area of the artificial habitat might otherwise develop. These findings reveal the multifunctional role of porous coastal defence structures, acting as engineering protection and habitats for coastal assemblages.

1. Introduction

Coastal areas provide essential economic resources and satisfy a variety of societal needs. Coastal ecosystems account for a substantial proportion of global ecosystem services (Costanza et al., 1999; Martínez et al., 2007), including coastal protection (Bulleri et al., 2005; Chapman and Underwood, 2011; Dugan et al., 2011; Garcia et al., 2004). Faced with the effects of accelerated climate change, coastal regions are susceptible to flooding and loss of land, requiring adaptational actions (Airoldi et al., 2005; Burcharth et al., 2007; Nicholls and Mimura, 1998; Philippart et al., 2011). The development of coastal defence structures (CDS) is fundamental in protecting land, property, infrastructure and other economic and environmental resources. Thus, in many areas worldwide, coastlines are becoming dominated by artificial structures (Airoldi et al., 2005; Bulleri and Airoldi, 2005; Firth et al., 2014; Firth et al., 2013a; Liquete et al., 2013; MAFF, 2000; Moschella et al., 2005) causing significant changes to shores through loss, replacement or fragmentation of natural habitats. This places intense pressure on coastal resources and the environment, and affects the structure and functioning of related marine ecosystems (Airoldi and Beck, 2007;

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Airoldi et al., 2005; Bulleri and Chapman, 2004; Connell and Glasby, 1999).

Infrastructure placed in any natural environment will inevitably become colonised by primary settlers such as epibenthic marine organisms and biofoulers (Evans, 2016). Artificial structures can be viewed as surrogate habitats for natural shores (Burt et al., 2011; Connell and Glasby, 1999; Moschella et al., 2005). With the aid of additional structural modifications to ameliorate habitat heterogeneity, increased colonisation and enhanced biodiversity of marine species on artificial substrates can be encouraged (Evans et al., 2016; Firth et al., 2013a, b, c). Currently, CDS are seen as poor substitutes for natural rocky shores because they support lower species diversity (Bulleri and Airoldi, 2005; Bulleri et al., 2005; Chapman and Blockley, 2009; Moschella et al., 2005). Coastal defence structures are typically built in high-energy environments with stronger wave action than most natural rocky shores (Burt et al., 2011; Evans et al., 2016; Jonsson et al., 2006), providing harsh habitat conditions for common rocky shore organisms, and opportunities for invasive non-native species through new hard substrata (Airoldi and Bulleri, 2011; Firth et al., 2013a). These conditions are made worse by scouring from sand, gravel and cobbles (Bulleri and Chapman, 2010; Moschella et al., 2005). Coastal defence structures are also less topographically complex than natural rocky shores, reducing habitat and microhabitat provision (Hawkins, 2012; Martins et al., 2010). Their extent is often smaller than natural shores (Moschella et al., 2005), inevitably leading to a restricted species pool and altered biological interactions amongst species (Bulleri and Chapman, 2010; Bulleri, 2005; Bulleri et al., 2005; Coombes et al., 2015; Jackson et al., 2008).

In conjunction with factors considered above, there is constant pressure on the structural integrity of CDS due to erosion, scouring, overtopping and undermining (Airoldi and Bulleri, 2011; Firth et al., 2013a; Kamphuis, 2010). Over time this can affect the stability and function of the structure, requiring maintenance (Airoldi, 2003; Dayton, 1971; Moschella et al., 2005; Sousa, 1979). Maintenance, however, can result in severe ecological disturbance. It can remove large areas of the habitat and causes disruption to settled communities by the abstraction and replacement of part or all of the structures (Tsinker, 2004; Airoldi and Bulleri, 2011). Such works can dislodge, crush or expose colonising species, potentially reduce biodiversity and open up space to opportunistic species (Dayton, 1971; Hutchinson and Williams, 2003; Sousa, 1979). Large costs are also incurred in the upkeep of the structures (Roebeling et al., 2011).

Porous rock defence structures are widely used in coastal engineering (Crossman et al., 2003). They serve a practical role in the protection of coastlines by reducing wave transmission, reflecting incident waves from the shores, and dissipating wave energy (Burcharth et al., 2015; Dalrymple et al., 1991; Garcia et al., 2004; Losada et al., 1995). Wave dampening is an important function that many other impermeable defence structures do not provide sufficiently (Garcia et al., 2004). The porous structure allows some of the wave energy to pass through whilst creating flow resistance and some reflection from the structure, resulting in turbulence through the porous medium and dissipation of wave energy (Garcia et al., 2004; Jung et al., 2012; Silva et al., 2000). Consequently, essential protection to the shoreline is provided whilst still allowing the natural process of water run-up on the coast. This imitates many natural shoreline barriers, such as coral reefs, mangroves and rocky shores, which can provide natural protection against waves and storm surges (Fernando et al., 2008; Hu et al., 2014; Lowe, 2005a, 2005b; Monismith, 2007).

Porous defence structures are also seen to be more environmentally friendly than solid CDS because they have a smaller physical footprint creating less disturbance to benthic soft sediment organisms (Koraim and Rageh, 2013), and can be more aesthetically pleasing (Garcia et al., 2004). Considerable recent work has focused on improving secondary functions of CDS, particularly enhancing their colonisation by marine biota. Research into artificial enhancements such as boring holes to create rock pools and drilled grooves to increase heterogeneity have been extensively researched (Borsje et al., 2011; Chapman and Blockley, 2009; Coombes et al., 2015; Evans et al., 2016; Firth et al., 2012, 2014, 2013, 2013b; Moschella et al., 2005; Naylor et al., 2011). Other studies have investigated the use of different materials to encourage settlement on the surface of these structures (Coombes et al., 2011a, 2011b, 2013; Green et al., 2012). Whilst this work has been a successful and an integral step towards working with nature by creating "green" infrastructure, the focus has been solely on the external surfaces of CDS. To date no work has been undertaken on the habitat provided by the internal surfaces of the rock units used in building porous CDS because of logistic constraints. Thus, this study presents the first opportunity to document the internal section of a porous rock armour structure. This is potentially a habitat providing some refuge from the harsh physical conditions of the intertidal zone in general (e.g. desiccation and wave action) and defence structures in particular (e.g. scouring).

The use of porous structures in coastal engineering can be viewed as providing a multifunctional role, protecting vulnerable coastlines and supporting intertidal communities. Our paper compares the community composition, abundance and biodiversity of species of internal versus external surfaces, taking advantage of the reduction of a groyne from 45 m to 20 m extent at Highcliffe on the South coast of the UK as part of reconfiguring an existing coastal defence scheme. More formally we tested the following hypothesis: internal habitats on the porous defence structure will support greater species richness and diversity than external habitats, in particular higher numbers of invertebrate species. In addition, we evaluate the extent of anthropogenic disturbance caused by the removal process, to indicate potential levels of general coastal defence maintenance disturbance and consider their possible impacts on coastal species.

2. Methodology

2.1. Study location

The study took place at Highcliffe in Christchurch Bay on the south coast of England, UK (Fig. 2.1), Christchurch Bay has a steadily eroding coastline of Barton clay beds and cliffs. It experiences a low amplitude double high tide, which is characteristic of the Solent area, meaning it encounters a further four tidal oscillations in addition to the standard semidiurnal UK tides. In spring tides the area experiences fluctuations in mean water levels of approximately 1 m (Nicholls, 1988; Tyhurst, 1986). There is also a complex tidal current system that circulates within the bay and a south-westerly wave pattern causing high-energy beaches to the west and local sediment drift and erosion. The area receives some protection from the Isle of Wight situated to the east and Durlston Head to the West (Tyhurst, 1986). The Highcliffe coastal defence scheme reverted from timber to rock groynes in 1992, and currently comprises eleven rubble mound groynes, consisting of short and long structures (30–45 m) and a bastion, made from Portland Oolitic limestone (Harlow, 2013; Tyhurst, 1986) (Fig. 2.2). The groynes are designed with 1 in 2 side slopes, 1 in 2.5 roundhead slopes and a 4 m crest width (Harlow, 2013). These are situated amongst a mixture of shingle and sand beaches (CBC, 2008), and the structures are estimated to sit approximately 1 m into the substrate. Christchurch Borough Council (CBC) deemed the groyne system at Highcliffe to be over engineered with a number of the groynes not being fully utilised within the coastal defence system. Therefore it was decided that the best approach was to remove and recycle the rock units. Owing to the direct attack from the sea, this area regularly undergoes routine maintenance work that consists of the replacement of rock units, removal/replacement of sand, or in some circumstances the partial reconstruction of a structure (CBC, 2008). The management of this area is essential to retain the current

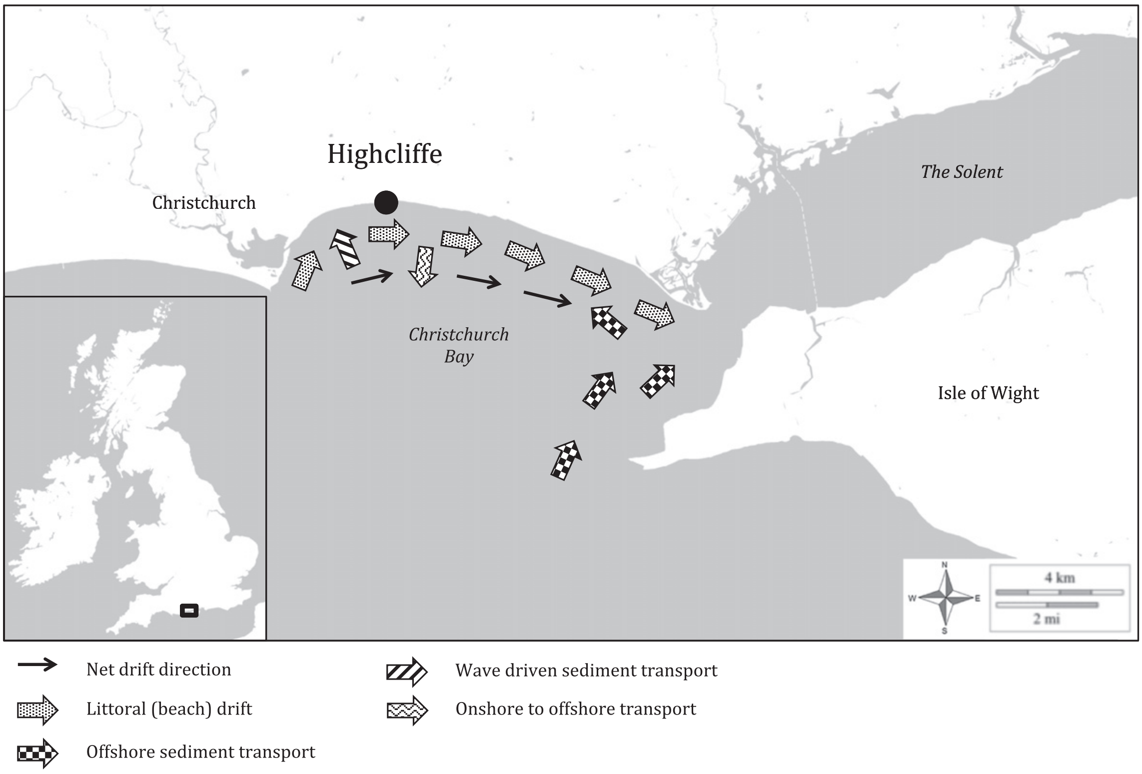

Fig. 2.1. Study area: Highcliffe situated within Christchurch Bay on the South coast of the UK. Map shows the sediment transport activity in the bay. Image adapted from MMIV © SCOPAC Marine Inputs map (http://www.scopac.org.uk/scopac_sedimentdb/chrst/index.htm).

coastline, protect residential properties and maintain the shoreline for tourism and local amenity use.

2.2. Groyne reduction

The groyne reduction took place during the lowest spring tides in June and July 2013 by CBC coastal engineers. The process removed 102 individual rock armour units of varying sizes (1–4 t rock units) roughly rectangular in shape, using a digger with a grab or bucket. The size of the structure was reduced from 45 m to approximately 20 m. Surface rock armour units from the end (nose) of the groyne were the first to be removed, exposing the foundation rocks. This allowed access to larger 4 t rock armour units that had sunk approximately 1 m into the sediment since they were installed. After the seaward nose and initial foundation units had been removed, the top and side layers were extracted followed by the internal (central) units. Owing to the short tidal window available for work, it was essential for the engineering work to be done in the specific removal order detailed above to ensure that structural integrity was retained between removal periods.

2.3. Data sampling

The restricted timeframes meant that ecological sampling was carried out around the engineering works; therefore all information was

Fig. 2.2. Image of the groyne system constructed at Highcliffe within Christchurch bay. Image shows the eleven rock groynes and a bastion of varying long and short lengths, and highlights the study groyne that was reduced. Image adapted from Imagery ©2016 Google, TerraMetrics, Map data ©2016 Google.

recorded in situ. Photographs and physical details of each unit were recorded, including measurements and calculations of the surface area of each unit face in order to determine the percentage cover species. Each unit face was recorded as an individual sampling point, and categorised by three different factors to determine the position and environmental exposure of each unit face (Fig. 2.3): (1) exposure to environmental conditions (comprising i, external wave exposed – outside unit face towards the seaward, ii, external wave sheltered – outside unit face with landward orientation, iii, internal – unit face located within the groyne, sediment – unit face located within soft sediment due sinking over time); (2) elevation on the shore (foundation – lower shore, middle, top of the shore); (3) placement of unit faces within the structure (nose – end of the structure, internal, side, top). Connections to other rock armour units and the estimated percentage damage to each unit from the removal process were also noted.

Biological sampling was conducted for each unit face by identifying organisms present to species level where possible with counts for mobile fauna and percentage cover for sessile species.

Maintenance disturbance was classified as areas of the unit face where fracturing and/or removal of the surface was visible due to the removal process. The level of maintenance disturbance was estimated by calculating the percentage of the unit face damaged or removed. The number of occurrences per rock armour face and frequency of occurrences out of the total sample were also logged.

2.4. Statistical analysis

To test our hypothesis, statistical analyses were carried out using PRIMER-E ver. 6 and PERMANOVA+ ver. 6 statistical software (Anderson et al., 2008; Clarke, 1993; Clarke et al., 2014) to determine the difference between species richness and percentage abundance of species recorded in relation to exposure levels. Moreover, we conducted supplementary analysis to determine if factors such as placement and elevation affect the data. Data were square-root transformed and a Bray-Curtis similarity matrix (Bray and Curtis, 1957) created for the statistical tests.

Multi-dimensional scaling (MDS) plots using abundance data were created for each factor (exposure, elevation on the shore and placement within the structure) to visualise patterns using rank similarities and hierarchical clustering in the multivariate output (Clarke, 1993; Clarke et al., 2014). Initial Permutational Multivariate Analysis of Variance (PERMANOVA), based on 999 unrestricted random permutations of residuals (Anderson et al., 2008), tested for differences in species richness and assemblages. Factors used in the analysis were: exposure (fixed, 4 levels: external, external sheltered, internal or sediment), elevation on the shore (fixed, 3 levels: foundation, middle or top), and placement within the structure (fixed, 4 levels: nose, internal, side or top unit). Pair-wise comparisons were used to test differences in the species richness and assemblages specifically in response to exposure levels. Similarity Percentage (SIMPER) was then used to identify percentage contributions of individual species providing the dissimilarities between the internal and external exposure levels (Anderson et al., 2008; Clarke, 1993; Clarke et al., 2014). Finally, we used Simpson's Index of Diversity (D) to calculate the species diversity in the internal and external (exposure levels) habitats.

For the maintenance disturbance, we calculated the average disturbance as a percentage of the surface cover, the number of occurrences and the extent of the damage as a percentage of the total number of samples for each factor level, to provide indicative data which may be used to inform methods for reducing disturbance levels.

3. Results

3.1. Ecological sampling

A total of 102 rock units were removed from the groyne structure, and the faces of each unit recorded. Species recorded during the removal process for the internal and external environments and, more

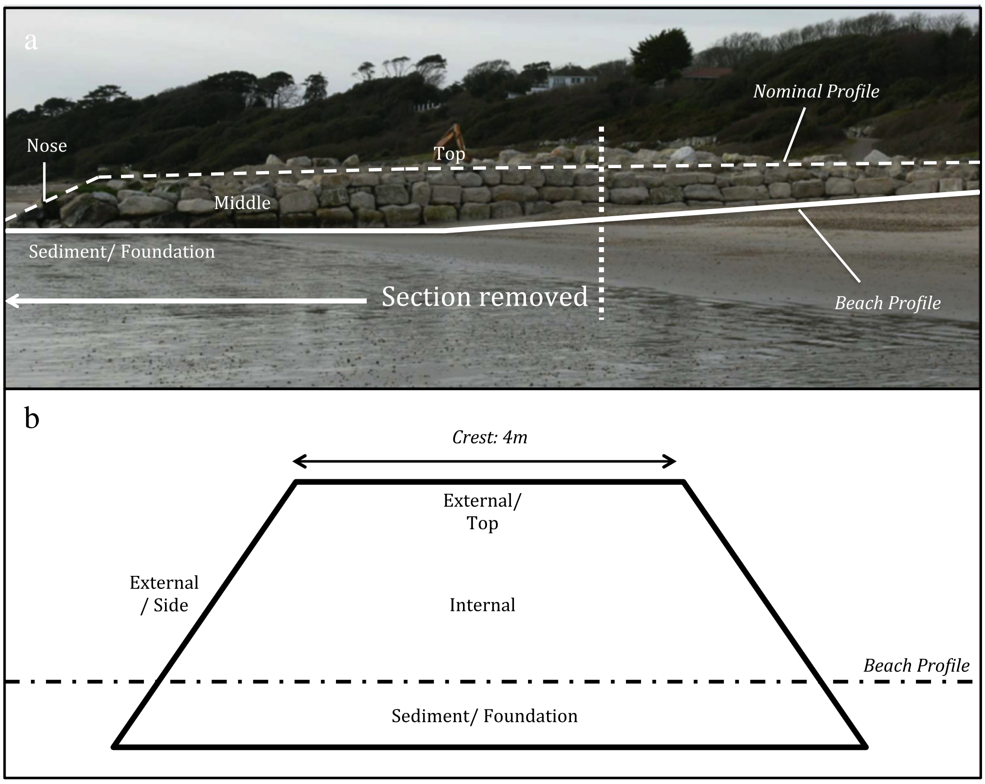

Fig. 2.3. Displays the areas categorised within each factor. (*a*) Shows a side view of the groyne that was removed, and the location of the categories under each factor that are visible. (*b*) Illustrates a landward facing cross-sectional representation of the groyne, and the relative locations of the categories under each factor.

Table 3.1
Total numbers of species and their mean percentage covers (sessile species only) in internal and external surfaces. Species diversity (*D*) was calculated using Simpson's Index. Number of unit sampled (n) = 102.

Group	Species	Internal				External			
		No. species	n	Mean % Cover	SD (±)	No. species	n	Mean % Cover	SD (±)
Green seaweeds		1				1			
	Ulva spp.		34	4.82	8.81		60	47.42	31.43
Red Seaweeds		6				2			
	Mastocarpus stellatus		15	0.38	1.12		15	0.30	0.73
	Chondrus crispus		28	7.87	15.89		4	1.33	5.21
	Hildenbrandia spp.		2	0.07	0.56		0	0.00	0.00
	Erythroglossum laciniatum		1	0.06	0.55		0	0.00	0.00
	Porphyra spp.		1	0.01	0.11		0	0.00	0.00
	Polysiphonia spp.		2	0.02	0.15		0	0.00	0.00
Brown Seaweeds		3				2			
	Fucus spiralis		6	0.48	3.32		9	0.66	2.39
	Algaozonia		1	0.24	2.18		0	0.00	0.00
	Dictyota dichotoma		0	0.00	0.00		3	0.13	0.58
	Sargassum muticum		5	1.87	10.26		0	0.00	0.00
Invertebrates		10				5			
	Patella vulgata		27	0.62	1.17		26	0.92	1.74
	Patella depressa		14	0.26	0.64		23	0.50	0.94
	Patella ulyssiponensis		7	0.12	0.45		8	0.22	0.65
	Cirripedia		30	6.88	13.75		12	1.64	5.82
	Actinia equina		19	0.35	0.88		0	0.00	0.00
	Actinia fragacea		1	0.01	0.11		0	0.00	0.00
	Mytilus edulis		35	0.70	1.40		15	0.31	0.75
	Eulalia viridis		3	–	–		0	–	–
	Carcinus maenas (juv)		1	–	–		0	–	–
	Nucella lapillus		4	–	–		0	–	–
Total		20	236	24.76	61.34	10	175	53.43	50.24
Simpson's Index of Diversity (D)		0.90				0.82			

specifically, the presence of species recorded on the rock unit faces and their percentage cover, are displayed in Table 3.1. Internal faces supported a higher number of species, particularly for invertebrate species and red seaweed species, than external unit faces (internal 20 species, external 10 species) (Table 3.1). Mobile fauna such as *Eulalia viridis,* juvenile *Carcinus maenas* and *Nucella lapillus* were all found only on internal faces. There was, however, a higher mean percentage cover of species found on external faces (53% ±50%), than on internal faces (25% ±61%) (Table 3.1). The results in Table 3.1 suggest that this is due to the presence of the alga *Ulva* spp., found in both environments, but more abundant on the external faces. *Ulva* spp. was recorded to cover on average 47% (±31%) of external faces compared to 5% (±9%) of internal faces. Calculations of Simpson's Index of Diversity (*D*) showed overall that internal surfaces had higher species diversity (0.90) than external surfaces (0.82).

An MDS plot (with 25% similarity contours) for exposure factors showed differences with exposure levels (Fig. 3.1*a*), particularly internal and external, where-as external sheltered and sediment levels appeared to be more distributed. Fig. 3.1*b & c* show no patterns with elevation on the shore and placement within the structure.

PERMANOVA analysis (Table 3.2) highlighted significant differences in the species assemblages due to exposure (*Pseudo-F* = 8.80, $P \leq 0.01$). Further analysis of the exposure factor using pair-wise tests showed significant differences between external areas and other exposure levels, particularly internal and external ($t = 5.20$, $P \leq 0.01$), and sediment and external ($t = 2.78$, $P \leq 0.01$). Analysis of the factors elevation and placement also showed significant differences in species assemblages. Additionally, PERMANOVA highlighted interactions between exposure and placement (*Pseudo-F* = 2.14, $P \leq 0.01$), and exposure and elevation (*Pseudo-F* = 2.16, $P \leq 0.01$), but no impact on species assemblages due to placement and elevation, or all factors combined. More specific analysis was not carried out, as this was not the focus of the study.

SIMPER analysis (Table 3.3) confirmed that *Ulva* spp. were the characterising organisms causing observed differences between internal and external faces contributing 42% of the dissimilarity observed. *Ulva* spp. were recorded on every external surface and covered the surface faces, whilst other species occupied smaller areas. *Chondrus crispus* (14%) and barnacles (13%) were contributing factors but were found in higher abundance on internal compared to external surfaces.

3.2. Maintenance disturbance

Table 3.4 shows that damage levels differed amongst locations. External units, alongside those located on the side of the structure and in the middle of the shore (see Fig. 2.3), had the highest incidences of maintenance damage and the highest average percentage cover per unit. External and external sheltered units had the highest number of occurrences as a proportion of the total sample, as well as those units located on the side and nose of the structure. Overall, maintenance disturbance recorded in the external units occurred much more frequently than internal units and caused a higher amount of damage to the surface.

4. Discussion

4.1. Habitat provision

The removal of a porous CDS provided a unique opportunity to gain better insight into the total habitat provision capabilities of artificial structures. It is unusual to come across the decommissioning of CDSs and this rare opportunity provided access to areas of artificial structures that have not previously been investigated or actively considered as a potential suitable habitat for coastal assemblages. By carrying out biological sampling during removal of a porous defence structure, we were able to gain important insights into the coastal species found on artificial structures.

We found significant differences between the biological communities present in internal and external environments (Table 3.1, Fig. 3.1*a*). Internal surfaces supported twice as many species of both

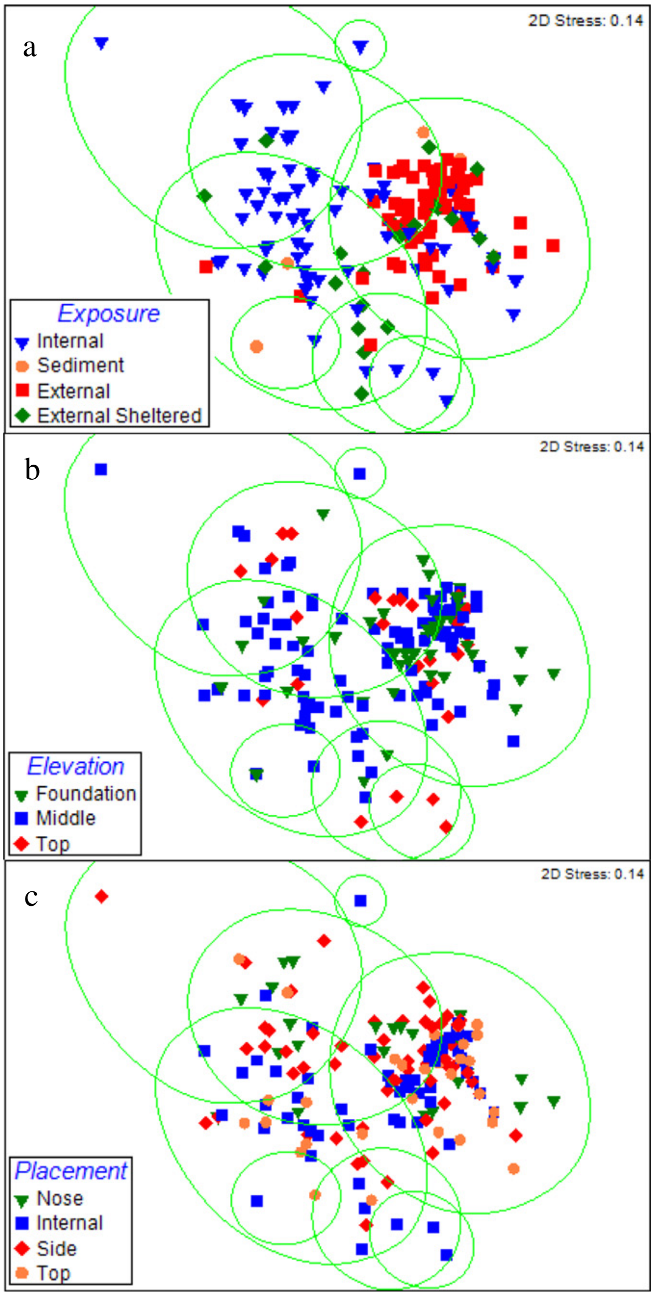

Fig. 3.1. MDS plots of percentage abundance data based on rank similarity for (*a*) exposure (*b*) elevation on the shore (*c*) placement within the structure (Fig. B). Contours of 25% similarity are shown.

invertebrates and algae as the external environment, particularly mobile species. A clear demonstration of the higher species richness associated with internal habitats on porous defence structures. Moreover, we found a greater species diversity overall on the internal than the external habitats. The results, however, showed higher total percentage cover by all combined species on external than internal habitats. Further analysis (Table 3.3) indicated *Ulva* spp. as the dominant species externally. *Ulva* spp. are green ephemeral opportunistic early successional species and require light to survive. They covered much of the rock unit surfaces in the external environment. This was most likely because of the unfavourable conditions for the majority of invertebrate species, and lack of grazing pressure (Coleman et al., 2006; Hawkins, 1981; Jenkins et al., 2005). The only species not recorded from the internal surface but present on the external exposed surface was

Table 3.2
PERMANOVA analysis identifying the impacts of the physical factors that affect community colonisation and species richness ($^{***}P(perm) = 0.001$;. $^{**}P(perm) < 0.01$; $^{*}P(perm) < 0.05$; NS = $P(perm) > 0.05$). Factors analysed were: exposure (fixed, 4 levels: external, external sheltered, internal or sediment), elevation on the shore (fixed, 3 levels: foundation, middle or top), and placement within the structure (fixed, 4 levels: nose, internal, side or top unit).

Factor	Interactions/ pair-wise test	Pseudo-F	df	t
Exposure		8.80^{***}	2	
	1. Sediment, External			$2\ 78^{***}$
	2. Sediment, External sheltered			1.34
	3. Sediment, Internal			1.00
	4. External, External sheltered			1.92^{*}
	5. External, Internal			5.20^{***}
	6. External sheltered, Internal			1.44
Placement		2.43^{**}	3	
Elevation		3.77^{***}	2	
Exp x Place		2.14^{***}	6	
Exp x Elev		2.16^{**}	4	
Place x Elev		1.44	3	
Exp x Place x Elev		1.11	3	

Dictyota dichotoma (brown fan weed). Invertebrate species were mainly found to colonise the internal areas of the structure in order to seek refuge to allow for foraging, whilst avoiding scour, wave exposure, desiccation and potential predation (Silva et al., 2008).

Despite the lack of visual relationships from the MDS plots, the overall results showed that there was a apparent difference in the species assemblages because of elevation on the shore and placement on the structure. The results also showed combined effects on species assemblages due to exposure and elevation on the shore, and exposure and placement on the structure. These are most likely because most external rock units will inevitably only be located on certain elevation or position areas such as the nose or sides of the groyne, compared to the internal exposure levels which would not be categorised under those locations.

The results of our study support our hypothesis that internal habitats on the porous defence structure will support greater species richness and diversity than external habitats, in particular higher numbers of invertebrate species. Coastal defence structures are constructed in high dynamic environments where there is increased pressure on invertebrate and plant species. *Ulva* spp. are known to colonise marine intertidal habitats (Bunker et al., 2010; Maggs et al., 2007) and dominate exposed surfaces leaving very little surface for other species to attach and colonise, therefore creating inter- and intraspecific competition for space and reducing biodiversity in these areas. *Ulva,* a green algae species, has high light requirements for photosynthesis, therefore colonising external, and non-shaded areas (Bunker et al., 2010; Maggs et al., 2007). *Chondrus crispus* and *Mastocarpus stellatus* successfully colonise the internal areas more than the external environments. Although these species often colonise exposed natural rocky shores, they can tolerate reduced light levels (sciaphilic) (Bunker et al., 2010)

Table 3.3
SIMPER analysis of the percentage (%) contribution of species to assemblage dissimilarities for and between exposure levels on the groyne (internal and external). Only species with contributions higher than 3% in at least one pairwise comparison are reported. Numbers in brackets are average dissimilarities between assemblages.

Species	Int x Ext	Internal	External
	(76.14)	(24.67)	(59.44)
Ulva spp.	42.72	70.39	5.57
Chondrus crispus	13.51	79.86	=
Barnacles	13.3	77.94	=
Patella vulgata	5.43	92.6	=
Mytilus edulis	5.01	88.61	=
Mastocarpus stellatus	3.44	=	=
Patella depressa	3.39	=	=
Actinia equina	3.38	–	–

Table 3.4
Average percentage cover of maintenance disturbance, number of occurrences of damage on internal and external rock armour units, and the frequency of occurrences (%) out of the total number of faces sampled ($n = 280$).

Factor	Level	Average Disturbance (% cover)	Number of occurrences	% of Total sample
Exposure				
	Sediment	0.0	0	0.0
	Internal	1.4 ± 6.0	8	7.3
	External sheltered	2.0 ± 6.2	5	20.8
	External	2.8 ± 8.2	15	14.2
Elevation				
	Foundation	0.5 ± 1.9	5	5.6
	Middle	2.5 ± 8.4	16	11.8
	Top	2.1 ± 6.2	7	12.7
Placement				
	Nose	2.8 ± 6.8	8	21.1
	Internal	0.8 ± 5.4	4	3.3
	Side	3.0 ± 8.4	13	15.5
	Top	1.1 ± 4.0	3	8.3

and are therefore able to colonise shaded, internal areas where there is more protection from waves. They are also later successional species and may be excluded by persistent ephemerals (Sousa, 1979). Invertebrate species were found primarily in the internal areas, with very few (low cover) exceptions. Species distributions on rocky shores are set by the interplay of vertical (tidal elevation) and horizontal (wave action) stress gradients, coupled with biological interactions (Raffaelli and Hawkins, 1996). Refuges are provided by microhabitats created by crevices, cracks and rock pools, which are common features of natural rocky shores (Johnson et al., 2003).

Until now, artificial structures have been perceived as poor surrogates for natural shores because they lack habitat complexity and heterogeneity (Chapman and Blockley, 2009; Firth et al., 2013a, b, c; Firth et al., 2013a; Moschella et al., 2005). Our study shows that porous defence structures do provide valuable habitats for species to colonise formed between rock unit interfaces providing refuge from desiccation stress (Hawkins and Hartnoll, 1983) and disturbance through scouring by cobbles, gravel and sand (Bulleri and Chapman, 2010; Moschella et al., 2005). Not only do porous defence structures effectively dissipate wave energy onto the coastline (Garcia et al., 2004; Jung et al., 2012; Silva et al., 2000), they also provide habitat complexity and protection within their interstices encouraging higher biodiversity than other types of coastal defence designs. They also enable water flow within the structure, providing access to food and submersion for periods, which is essential for many intertidal species. The multifunctionality of porous defence structures is clearly a desirable feature and the benefits conferred are valuable considerations which may usefully inform both engineering and management.

4.2. Maintenance disturbance

There has been little research investigating the effects of maintenance disturbance on coastal assemblages (but see Airoldi and Bulleri, 2011). Our study demonstrates the levels of disturbance that occur during coastal maintenance, particularly to internal and external environments. Anthropogenic disturbance can create openings for opportunistic and invasive non-native species to settle (Dayton, 1971; Hutchinson and Williams, 2003; Sousa, 1979). One key finding is the difference in disturbance levels between internal and external environments. There was nearly double the amount of maintenance disturbance on the external rock unit faces compared to internal ones. Typical coastal maintenance will often involve the replacement of a number of rock units that may become dislodged or moved during intense weather conditions. This will mainly be on external rock units that are more exposed to the extreme conditions and susceptible to movement. Moving units during maintenance work to restore structural integrity after storm damage is an activity that will disrupt those species occupying affected units, as well as species associated with any connecting units. This emphasises the importance of internal environments as suitable habitats to support higher levels of biodiversity on coastal shores. Future work should be carried out to investigate further the effects of disturbance.

5. Concluding comments

Until now, the internal environment of CDS has not been actively considered or explored by ecologists for its potential to provide habitat and enhance biodiversity. Our study highlights the importance of these hidden environments for coastal species, suggesting that porous CDS provide improved habitat heterogeneity and refuges via internal compartments. These features are not present in solid structure designs with no internal compartments. External environments on coastal defence structures are exposed to intense environmental pressures made worse by anthropogenic disturbance from any maintenance work. Therefore they only support a small number of hardy species. Focus must be turned to the internal environment, which can support a higher diversity of species. Porous structures, a common coastal engineering design, are not only effective in engineering; they are also considerably more effective for biodiversity than previously realised. Porous CDS should be considered more widely in future coastal engineering schemes, to encourage settlement of coastal species and to sustain coastal communities, particularly given the growing number of artificial structures and in light of gross environmental change and habitat loss. Finally, further investigations into the impacts of maintenance activity on coastal assemblages should be considered to inform coastal engineers and to provide evidence-based decisions for effective coastal defence management regimes.

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Maintenance activity and the hidden habitats in coastal defence structures: new steps towards effective ecological management

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Introduction

Coastal areas are essential for both economic resources and societal needs. Coastal ecosystems account for a substantial proportion of global ecosystem services (Costanza *et al.*, 1999; Martínez *et al.*, 2007), including coastal protection. Faced with the effects of accelerated climate change, coastal regions will be susceptible to flooding and loss of land, requiring mitigating actions (Nicholls and Mimura, 1998; Airoldi *et al.*, 2005; Burcharth *et al.*, 2007; Philippart *et al.*, 2011). The development of coastal defence structures (CDS) is fundamental in securing land, property infrastructure and other economic resources. Thus, in many areas worldwide, coastlines are becoming dominated by artificial structures (MAFF, 2000; Moschella *et al.*, 2005; Liquete *et al.*, 2013) putting intense pressure on coastal resources and the natural environment. This results in significant changes to coastlines through loss, replacement or fragmentation of natural habitats, and is affecting the structure and function of the marine ecosystems (Connell and Glasby, 1999; Bulleri and Chapman, 2004; Airoldi *et al.*, 2005; Airoldi and Beck, 2007; Zanuttigh *et al.*, 2010; Chapman and Underwood, 2011; Firth *et al.*, 2013a).

Artificial structures can become surrogate habitats for natural shores (Connell and Glasby, 1999; Bulleri, 2005; Moschella *et al.*, 2005; Burt *et al.*, 2009; Burt *et al.*, 2011). With the aid of additional structural enhancements to increase habitat heterogeneity, colonisation of coastal species on artificial substrates can be encouraged (Firth *et al.*, 2013a; 2013b). Artificial habitats are however, currently seen as poor replacements for natural shores. This is because of their placement in harsh wave exposed conditions, made worse by scouring from sand, gravel and cobbles. Defence are less topographically complex than natural rocky shores, reducing habitat and microhabitat diversity. Their extent is often smaller than natural shores, inevitably leading to a smaller species population. Species colonising these areas are already subject to higher environmental stress as coastal defence structures are generally implemented in areas where there is a demand for shoreline protection. Therefore the natural storm conditions are stronger and more frequent than in other more sheltered coastal areas (Jonson *et* al., 2006; Burt *et al.*, 2011). In addition, constant pressure on defence structures due to erosion, scouring, overtopping, undermining and other damages (Kamphuis, 2010; Airoldi and Bulleri, 2011; Firth *et al.*, 2013b). Overtime this can affect the stability and function of the structure, requiring maintenance (Airoldi, 2003; Moschella *et al.*, 2005). Maintenance, however, is an exceptionally severe form of ecological disturbance. It can remove large areas of the habitat and causes disruption to settled communities by the removal and replacement of part or all of the structures (Dayton, 1971; Sousa, 1979; Tsinker, 2004; Airoldi and Bulleri, 2011). This activity can dislodge, crush or expose species that colonise the shore, potentially reduce biodiversity, open up space to opportunistic species (Dayton, 1971; Sousa, 1979; Hutchinson and Williams, 2003), and incur large costs attached to the upkeep of the structures. The environmental repercussions from maintenance work means that a full understanding of the impacts of anthropogenic disturbance is essential in order to minimise disruption, and to design sustainable coastal schemes. Maintenance work often consists of the replacement of rock units, removal/replacement of sand, or in some circumstances the partial reconstruction of a structure. There are, however, a number of cases where full rock armour structures have been redesigned and replaced because the original design did not meet the local area requirements or conform to updated guidelines. There is very little understanding of the environmental impacts from maintenance activity on biological communities on or in the surrounding areas of coastal defence structures.

Considerable recent work has been done on colonisation by marine biota of the outside surfaces of sea defence structures. Most structures are however, porous and to date no work has been done on the habitat provided by the internal surfaces of the blocks used in building structures. This is potentially a habitat providing refuge from the harsh physical conditions of the intertidal zone in general (e.g. desiccation and wave action) and defence structures in particular (e.g. scouring). We took advantage of the removal of a groyne at Highcliffe on the South coast of the UK to compare biodiversity of internal versus external surfaces and we explored the impacts of maintenance and removal of structures on coastal assemblages. This is the first time the internal section of a porous rock armour structure has been studied. We compare the community composition, abundance and diversity of species in different habitats on a coastal defence structure.

Methodology

Study location

The study took place at Highcliffe in Christchurch Bay on the South coast of the UK. Christchurch Bay has a steadily eroding coastline of Barton clay beds and cliffs. It experiences a double high tide, which is characteristic of the Solent area. There is also a complex tidal system that circulates within the bay and a south-westerly wave pattern causing high-energy beaches to the west and local sediment drift and erosion. The area benefits from some protection from the Isle of Wight situated to the east and Durlston Head to the West (Tyhurst, 1986). The Highcliffe coastal defence currently consists of eleven rubble mound groynes, consisting of short and long structures (60-80m) and a bastion (Tyhurst, 1986). Christchurch Borough Council (CBC) deemed the groyne system at Highcliffe to be over engineered with a number of the groynes not being fully utilised within the coastal defence system. Therefore it was decided that the best approach was to remove and recycle the rock units. The management of this area is essential to retain the current coastline and economic assets that run along the coast in Christchurch, mostly residential properties and tourism.

Groyne removal

The groyne reduction took place in June and July 2013 working with CBC coastal engineers. The process removed individual rock armour units of varying sizes (1-4+ tonne rock units), using a digger and grab or bucket. The structure size was reduced to about 20m (1/3rd of its original size). Nose rock armour units at the surface were the first to be removed and exposed the foundation units at the end of the groyne. This allowed access to larger 4+ tonne rock armour units that had sunk into the sediment since they were installed. After the nose and foundation units had been removed, the top and side layers were extracted followed by the internal units. Photographs and physical details of each unit were taken, along with the estimated size of each unit face and estimated weight of the unit. Each unit face was categorised by different factors: placement on the shore (foundation, middle, top of the shore); placement within the structure (nose, internal or side unit); and exposure level of each unit face (exposed, sheltered, internal or sediment). Connections to other rock armour units and the percentage damage to each unit from the removal process were also noted. Ecological sampling of percentage cover of coastal species was recorded for each unit face and species were identified to species level where possible. After units were removed, they were placed higher up the shore to be recycled in future defence schemes.

Statistical analysis was carried out to determine the difference between community colonisation, abundance and biodiversity in relation to location on the groyne. Permutational Multivariate Analysis of Variance (PERMANOVA) was conducted using Bray-Curtis similarity coefficients, calculated using fourth-root transformed data to stabilise variances (Anderson *et al*., 2008). For the analysis, 999 unrestricted random permutations of residuals were used to generate *P-values*. The analysis included the factors: exposure (fixed, 4 levels: exposed, sheltered, internal or sediment), placement on the shore (fixed, 3 levels: foundation, middle or top of the shore), and placement within the structure (fixed, 3 levels: nose, internal or side unit). A non-metric multi-dimensional scaling (MDS) plot, calculated on a Bray-Curtis similarity matrix for each combination of exposure, placement on the shore and within the structure, was used to visualise patterns and similarities in multivariate data.

Results

Using position on the structure, PERMANOVA analysis of the interactions between factors was explored (Table 1). The results showed significant differences in the assemblages due to exposure, placement of the unit on the shore (base) and within the structure (placement) (PERMANOVA for exposure *Pseudo-F* = 8.8004, P = 0.001; placement on the shore *Pseudo-F* = 3.7672, P = 0.001; placement within the structure *Pseudo-F* = 2.4388, P = 0.004). Furthermore, the PERMANOVA analysis highlighted important differences between the exposure and placement, and exposure and base (PERMANOVA for exposure and placement *Pseudo-F* = 2.1426, P = 0.001; exposure and base *Pseudo-F* = 2.163, P = 0.003).

Table 1 Results from PERMANOVA analysis identifying the impacts of the physical factors that effect community colonisation (significant values in bold).

		PERMANOVA			
Factor	Interactions/ Pair-wise test	*Pseudo-F*	*P*	*t*	*P*
Exposure		**8.8004**	**0.001**		
	1. Sediment, External			**2.7782**	**0.001**
	2. Sediment, External sheltered			1.3379	1.149
	3. Sediment, Internal			0.9979	0.43
	4. External, External sheltered			**1.9193**	**0.031**
	5. External, Internal			**5.2042**	**0.001**
	6. External sheltered, Internal			1.441	0.061
Placement		**2.4388**	**0.004**		
Base		**3.7672**	**0.001**		
Exp x Place		**2.1426**	**0.001**		
Exp x Base		**2.163**	**0.003**		
Place x Base		1.4428	0.154		
Exp x Place x Base		1.1129	0.335		

MDS plots shown in Figure 1 further explore the results by spatially presenting the community data and highlighting trends or relationships within the data. The MDS plot for exposure display a noticeable separation between the external and internal assemblages. However, the MDS plots for the placement on the shore and within the structure did not display any discernable trends in the data despite the results from the PERMANOVA analysis (but has been included for comparative purposes). The MDS plot for placement display a slight trend in the communities found on the nose of the structure as they tend to be presented in the top part of the plot, but there is still no distinct cluster formation, and much overlapping between the data points for each factor level. Following the results from the PERMANOVA and MDS plots; further analysis using Pair-wise tests indicates the effects of exposure on the coastal communities. The analysis revealed substantial differences between external areas compared to internal sections (PERMANOVA for exposed vs. internal areas t = 5.2042, P = 0.001). The other exposure levels did not show any significant results from the analysis.

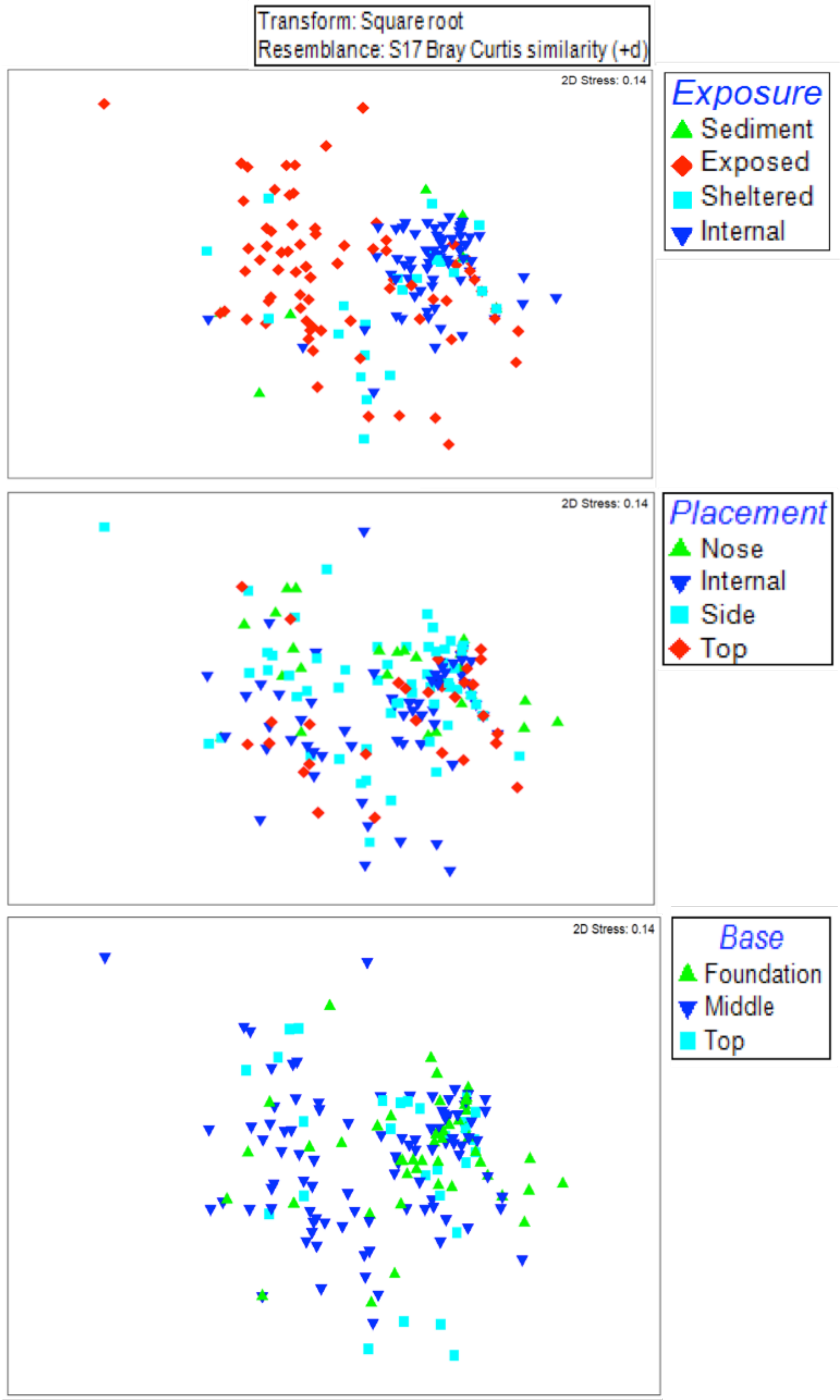

Figure 1 Multi-dimensional Scaling plots spatially presenting interactions between data for each unit face and the factors affecting colonisation.

In addition, data collected for the level of maintenance disturbance found in Table 2, highlights the maintenance disturbance for each unit, and the count of disturbance events on rock armour faces. There are three distinct outcomes in the results in Table 2 that show the highest number of occurrences where maintenance disturbance was observed and the highest average percentage cover per unit. These are external exposure, side units on the structure and from the middle of the shore.

Maintenance disturbance recorded in the external units are occur more frequently and cause a higher amount of damage to the surface.

Table 2 Average percentage cover maintenance disturbance and number of occurrences on internal and external rock armour units.

Factor	Level	Average Percentage Cover (%)	Number of Occurrences
Exposure			
	Sediment	0	0
	Internal	1.36 (±5.98)	8
	External sheltered	1.96 (±6.21)	5
	External	2.81 (±8.19)	15
Placement			
	Nose	2.76 (±6.75)	8
	Internal	0.84 (±5.44)	4
	Side	2.95 (±8.39)	13
	Top	1.11 (±3.98)	3
Base			
	Foundation	0.45 (±1.94)	5
	Middle	2.50 (±8.36)	16
	Top	2.09 (±6.21)	7

Analysis was carried out on the number of species recorded and their percentage cover on internal and external faces of rock armour units. Table 3 shows that there was a higher number of species found in the internal environment (18 species) than the external (9 species). There was, however, a higher mean percentage cover of species found on external faces (53.48% (±50.75%)), than on internal faces (24.86 (±61.84)). Additionally, there was one dominant genus, *Ulva* spp., found in both environments, was more abundant on the external faces. *Ulva* spp. was recorded to cover on average 47.42% (±31.43%) of external faces compared to 4.82% (±8.81%) of internal faces.

Table 3 Mean percentage cover and total number of species found in internal and external environments

		Mean Percentage Cover (%)		Total Number of Species	
Group	Species	Internal	External	Internal	External
Green Seaweeds				1	1
	Ulva spp.	4.82 (±8.81)	47.42 (±31.43)		
Red Seaweeds				6	2
	Mastocarpus stellatus	0.38 (±1.12)	0.30 (±0.73)		
	Chondrus crispus	7.87 (±15.89)	1.33 (±5.21)		
	Hildenbrandia spp.	0.07 (±0.56)	0		
	Erythroglossum laciniatum	0.06 (0.55)	0		
	Porphyra spp.	0.01 (±0.12)	0		
	Polysiphonia spp.	0.02 (±0.15)	0		
Brown Seaweeds				3	2
	Fucus spiralis	0.48 (±3.32)	0.66 (±2.39)		
	Algaozonia	0.24 (±2.18)	0		
	Dictyota	0	0.13 (±0.58)		

	dichotoma				
	Sargassum muticum	1.87 (±10.26)	0		
Invertebrates				8	4
	Patella vulgate	0.62 (±1.17)	0.92 (±1.74)		
	Patella depressa	0.26 (±0.64)	0.5 (±0.94)		
	Patella aspera	0.12 (±0.45)	0.22 (±0.65)		
	Cirripedia	6.88 (±13.75)	1.64 (±5.82)		
	Actinia equine	0.35 (±0.88)	0		
	Actinia fragacea	0.01 (±0.11)	0		
	Eulalia viridis	0.036 (±0.19)	0		
	Carncinus maenas (juv)	0.01 (±0.11)	0		
Total		24.86 (±61.84)	53.48 (±50.75)	18	9

Discussion

Carrying out biological and physical sampling during removal of a porous defence structure provided an opportunity to explore the coastal communities found in all environments on an artificial structure, and to gain insight into the impacts of maintenance activity on these assemblages. This study highlights important results about the coastal species found on artificial structure, and provides valuable insight into the levels of anthropogenic disturbance.

We found significant differences between the biological communities present in internal and external environments due to exposure level (Table 1 and Figure 1). Internal environments accounted for twice as many species as the external environment (Table 3) but supported mainly invertebrate species, compared to the external surfaces where seaweeds dominated. The results also showed higher percentage cover of species in external environments than internal. This was due to the presence of one dominant species in the external environment; *Ulva* spp. *Ulva* spp. is an ephemeral opportunistic species and was observed during the removal process to cover huge amounts of the rock armour surfaces in the external exposed environment. This is most likely because of the unfavourable conditions for most invertebrates' species, and therefore a lack of grazing pressure.

The results show significant differences in species found in different areas of the shore and an association between the species found due to exposure level and placement within the structure. These results would be expected due to the biological requirements of invertebrate and plant species. Seaweeds require exposure to sunlight in order to photosynthesis; however, invertebrate species do not and tend to retreat into sheltered areas which provide protection from desiccation and predators. Different species also respond to fluctuating sea levels and environmental pressures such as wind exposure, periods of submersion, competition, and predation (Haslett, 2000; Jackson and McIlvenny, 2011). The relationship between exposure levels and placement on the shore should correspond to the fact that some rock armour units exposed to external environments will most likely be found on the sides or nose of the groyne, compared to the internal exposure levels which should relate to the internal units.

We suggest that internal environments provide a more suitable, alternative habitat for coastal assemblages than the exposed areas. This is not surprising when considering the refuge provided by internal environments compared to external, and the niche microhabitats formed crevices between rock armour unit interfaces that many species rely on (Haslett, 2000; Johnson *et* al., 2003; Jackson and McIlvenny, 2011). Porous defence structures will also enable water flow within the structure, providing access to food and submersion for periods essential for many intertidal species.

There has been little research to date carried out investigating the effects of maintenance disturbance on coastal assemblages (see Airoldi and Bulleri, 2011). This study acknowledges the levels of disturbance that occurs during coastal maintenance, particularly internal and external environments. Anthropogenic disturbance can create disruptions to the dynamics of the coastal communities and openings for opportunistic and invasive species to settle (Dayton, 1971; Sousa, 1979; Hutchinson and Williams, 2003). Our key finding is the difference in disturbance levels between internal and external environments. There is nearly double the amount of maintenance disturbance that occurs on the external rock unit faces compared to internal ones. Future work should be carried out to further investigate the effects of disturbance. This provides supplementary evidence indicating that internal environments are more suitable habitats to support higher levels of biodiversity on coastal shores. Typical coastal maintenance will often involve the replacement of a number of rock units that may become dislodged or moved during intense weather conditions. This will mainly be on external rock units that are more exposed to the extreme conditions and susceptible to movement. This not only creates potential weakness to the structure and its function, but also disrupts species occupying the units moved and any associated connecting units. The internal environment however, is protected from this disturbance.

Conclusions

Until now, the internal environment of coastal defence structures has not been explored or considered by engineers and ecologists, for its potential to encourage and sustain coastal assemblages. This study pinpoints the importance of these hidden environments for coastal species, suggesting that coastal defence structures can become suitable surrogate habitats if we focus on optimising the internal compartments of defence structures so they provide functional habitats as well as shoreline protection. External environments on coastal defence structures are exposed to intense environmental pressures alongside anthropogenic disturbance from maintenance work. Therefore they only support a small number of hardy species. Focus must be turned to the internal environment, which can support a higher biodiversity of species. Porous structures are not only effective in engineering; they are also effective for biodiversity. It is imperative that we optimise coastal defence designs and management schemes with this new evidence in mind, to try to encourage settlement and sustain our natural coastal communities. It is also essential to further investigate the impacts of maintenance activity on coastal species so we can provide evidence-based decisions for effective management schemes.

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