The Ecology and Ecological Enhancement of Artificial Coastal Structures

By

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A thesis submitted in partial fulfilment of the requirements of
Bournemouth University for the degree of Doctor of Philosophy

Bournemouth University

November 2017

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Abstract

Due to predicted sea level rise, increased storm frequency and coastal urbanisation, artificial coastal structures are proliferating worldwide. It is now vital to understand the impact that artificial structures are having on the marine environment and it is becoming increasingly necessary to incorporate ecologically enhanced designs into both new and existing coastal infrastructure. This thesis examines the communities associated with coastal artificial structures and trials ways in which we can improve the habitat provided for marine organisms. Firstly, the existing communities present on wooden and rock groynes within four regions along the south coast of England were surveyed and quantified and compared to natural rocky shores. The groynes were found to be less diverse than the natural rocky shores yet the wooden groynes supported a greater variety of species compared to the rock groynes. Secondly in order to assess the stability of individual populations, the variation in recruitment and mortality of the barnacle *Semibalanus balanoides*, and recruitment of the limpet *Patella vulgata* and the mussel *Mytilus edulis* were monitored on wooden and rock groynes within two regions and compared to natural rocky shores. High spatial variation in recruitment was observed, yet overall, communities on the groynes were comparably stable to those on the natural rocky shores. This thesis also investigates the impacts of artificial structures on surrounding mobile communities and their contribution into the food web. Baited Remote Underwater Video (BRUV) and Stable Isotope Analysis (SIA) were used to assess the distribution of mobile fauna around pleasure piers and to determine if the mobile fauna were using the pier as a feeding resource. The results showed a greater number of species in close proximity to the pier and the SIA found evidence of links between trophic levels of species collected on and around the piers. In order to assess the connectivity between populations, hydrodynamic and particle tracking models were used to determine the potential for artificial structures to act as stepping stones for the dispersal of native and non-native species. As the majority of artificial structures are known to lack biological diversity, this thesis explores how increasing the surface heterogeneity of rock armour using low cost enhancement options and improving water retention on seawalls by attaching artificial rock pools can improve diversity. The ecological enhancements attracted new species including mobile fauna which were previously absent and increased the overall diversity of the structures. To conclude the key findings are summarised and future work and management advice is outlined.

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Acknowledgements

Firstly I would like to thank my supervisors, Dr Roger Herbert, Prof Rob Britton and Dr Ben Thomas for their guidance and support throughout my PhD. I am very grateful for all of our discussions and for allowing me to develop my own project, yet still making sure it was achievable. I have thoroughly enjoyed my PhD and I strongly believe it is down to such valuable supervision, so thank you. I would also like to thank John Baugh from HR Wallingford who has helped and advised me in the creation of the hydrodynamic model for Poole Bay. Alongside, Jay Willis from Oxford University to who I owe a huge amount, for helping me develop the larvae dispersal models without whom it would have been impossible- thank you for teaching me the delights of Matlab. I would also like to thank Arc Consulting, Artecology and Eccelston George (Clare, Dave, Ian, Nigel, Sam) for both the opportunity to monitor the Vertipools but also for the stimulating discussions we have had, you are all truly inspiring people and I look forward to working with you in the future.

Thank you to all my field assistants whom have helped me over the last three years; Ailidh Barnes, Alex Lovegrove, Ann Thornton, Ben Thornes, Clare Davies, Danny Sheath, Dave Hartnell, Dean Burnard, Hayley Roberts, John Atkinson, Leo Clarke, Neil Bradley, Richard Mikulski, Tea Basic, Su Chern Foo, Wayne O'Brien. From scraping barnacles off rocks in the pouring rain to seine netting in the sunshine I could not have done it without your help (even if I did have to bribe some of you with ice creams). I would also like to thank the local anglers who helped collect my fish samples from around the piers. In addition I would like to thank John Humphreys from JHConsulting for his excellent boating assistance and Lin Baldock for species identification help.

I would like to thank Dave Harlow from Bournemouth Borough Council and Dave Robson from Poole Borough Council for the information on the groynes in Region 1, Andrew Pearce from the Eastern Solent Coastal Partnership who supplied information on the Region 2 groynes and Adur & Worthing Council Engineering Services who supplied the data on the Region 3 groynes. I would also like to thank the National Trust for allowing me assess on Brownsea Island to complete my fieldwork surveys.

A PhD is certainly a rollercoaster and I would not have come out the other side if it weren’t for all my friends and family so thank you all very much for your support. Special thanks to my mother and my in laws who have given up their holidays to help me with fieldwork! I would also like to thank Dr Sue Hull from University of Hull for her continued support and encouragement in helping me become the academic I am today.

Finally, most importantly, I need to thank my husband Lee without whom I would not have been able to survive the last few years. He has helped with fieldwork, listened to me rant on about on matters he did not understand and been there to support me through the difficult times, I will be eternally grateful. One lesson I have learnt is to appreciate how much I have accomplished in the last three years and the best achievement in life is doing something you thought you could never do!

The Ecology & Ecological Enhancement of Artificial Coastal Structures

1 General Introduction

1.1 Ocean sprawl

Currently over 40% of the world population lives in coastal areas (<150 km from the sea) and this is set to increase in future years (Cohen, 1997; Nicholls et al., 2007). Coupled with the impacts of climate change such as sea level rise and increased storm frequency, the number of coastal artificial structures is proliferating around the world (Pethick, 2001; Wang et al., 2012; Hinkel et al., 2014). ‘Ocean sprawl’ is the increase in the number of artificial structures associated with coastal and offshore areas (Dafforn et al., 2015; Firth et al., 2016a; Bishop et al., 2017; Heery et al., 2017). Over 50% of parts of coastlines of Europe, USA, Australia and Asia have been modified by hard engineering which has predominantly been for coastal defence structures such as seawalls, breakwaters and groynes (Bacchiocchi & Airoldi, 2003; Moschella et al., 2005; Vaselli et al., 2008; Firth et al., 2013a). Other hard engineering includes of the construction of ports, harbours and infrastructure for industrial activities such as oil rigs (Parente et al., 2006), renewable energy devices (Witt et al., 2012) and underwater pipelines (Dafforn et al., 2015). This thesis focuses on improving the understanding of the ecology of artificial structures in order to enhance biological communities associated with structures and prevent the loss of biodiversity.

1.2 Artificial coastal structures

Hard artificial structures typically consist of man-made materials designed and constructed to perform a specific purpose (Mineur et al., 2012). The various forms of artificial structures result in changes to substrate type, surface texture and complexity which can create a range of novel habitats within the marine environment. Impacts of artificial structures can be i) direct physical disturbance from the addition of materials during construction, ii) addition of artificial habitat, altering the connectivity of structures and habitat composition, iii) indirect physical disturbance, through changes in sediment transportation and altered turbidity, iv) noise and light pollution (Dafforn et al., 2015). In addition, artificial structures can alter the biological interactions, trophic interactions and functioning of an ecosystem (Bishop et al., 2017). These impacts can occur on both a local and regional scale which can lead to an overall reduction in biodiversity.

1.2.1 The ecology of artificial coastal structures

Hard substrates are typically colonised by sessile species such as barnacles, mussels, bryozoans, hydroids and macroalgae (Bacchiocchi & Airoldi, 2003; Bulleri & Chapman, 2004; Mineur et al., 2012), which in turn creates biogenic habitats for additional fauna including mobile species such as crustaceans, fish and cephalopods (Coleman & Connell, 2006; Wilhelmsson et al., 2006; Clynick et al., 2007). However benthic intertidal assemblages of sessile species associated with artificial structures have been found to be less diverse than natural rocky shore communities (Bacchiocchi & Airoldi, 2003; Chapman & Bulleri, 2003; Bulleri & Chapman, 2004, 2010; Chapman, 2006; Underwood & Chapman, 2006; Vaselli et al., 2008; Firth et al., 2013b). Artificial structures have also shown to support different assemblages to natural rocky shores (Chapman & Bulleri, 2003; Moschella et al., 2005; Pinn et al., 2005) with reduced diversity of mobile species (Chapman, 2003; Pister, 2009; Aguilera et al., 2014) and a lower genetic diversity (Fauvelot et al., 2009) .

1.2.2 Factors affecting communities on coastal artificial structures

Various factors influence the community composition on coastal artificial habitats. These can be split into two categories: firstly biophysical and hydrodynamic factors and secondly substrate factors. Biophysical and hydrodynamic factors include the influence of larval supply, tidal height, depth and wave exposure whereas substrate factors include substrate type, structural complexity, age, enhancements and maintenance (Figure 1.1).

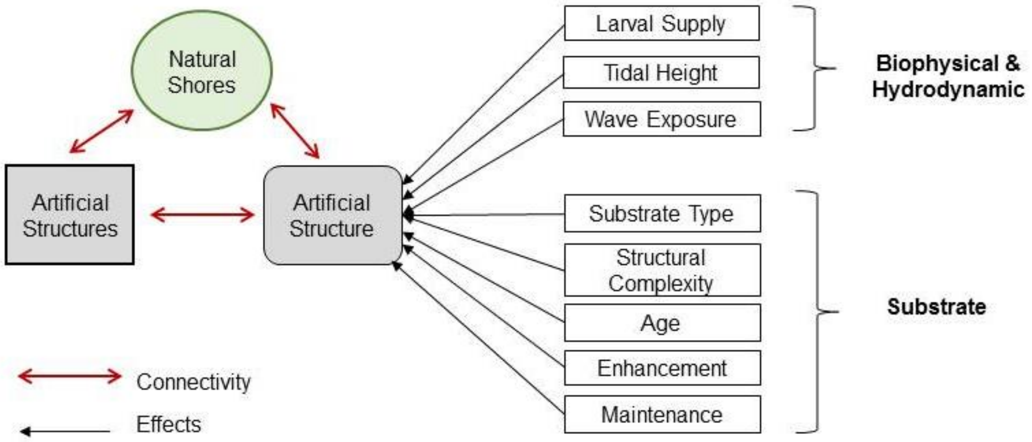

Figure 1.1 Conceptual diagram showing the major factors affecting the biodiversity of coastal artificial structures. Arrows between the different habitats indicate potential connectivity via propagule supply

Coastal defence structures are predominantly constructed above the mean low water mark in order to protect the land from erosion (French, 2001; Burcharth et al., 2007). The higher the tidal height the greater the chance of disturbance from sand scour (Moschella et al., 2005), wave energy (Firth et al., 2013a) and desiccation stress. Coastal defence structures are largely built in high wave energy environments which in turn limits the numbers of species adapted to survive the harsh conditions (Moschella et al., 2005; Burcharth et al., 2007; Firth et al., 2014b).

The habitat complexity provided by a structure has a large impact on the community composition. Studies have found that the reduced species diversity on artificial structures in comparison to natural shores is due to the lack of habitat complexity (Chapman & Bulleri, 2003; Moschella et al., 2005; Evans et al., 2017). The majority of artificial structures are built from materials such as concrete or metal and finished with smooth surfaces, which creates a poor habitat for colonisation (Chapman, 2006; Firth et al., 2013b; Browne & Chapman, 2014). Artificial structures lack the water retaining features commonly found on natural rocky shores in the form of rock pools (Moschella et al., 2005; Chapman & Blockley, 2009; Bulleri & Chapman, 2010; Firth et al., 2013b) and microhabitats which provide vital refuges for intertidal organisms (Chapman & Blockley, 2009). Intertidal refuges provide vital escapes from harsh environmental conditions such as extremes of temperature and desiccation and also biological factors such as competition and predation (Fairweather, 1988; Metaxas & Scheibling, 1993; Johnson et al., 1998). Increased areas of refuges have shown to increase the biodiversity on both natural rocky shores (Chapman & Blockley, 2009; Firth et al., 2013b, 2014a) and on artificial structures (Firth et al., 2013b; Evans et al., 2015).

In terms of age, the benthic communities found on artificial structures are generally much younger than those on natural rocky shores, with the exception of highly disturbed natural shores (Connell & Glasby, 1999). Natural shores have been present in the marine environment for thousands of years and have undergone various erosional processes which have formed complex features such as rock pools and crevices (Coombes, 2011) which creates dynamic habitats (Raffaelli & Hawkins, 1996). Natural disturbance events occur on natural rocky shores (Little et al., 2009) and artificial structures, yet artificial structures can be particularly prone to disturbance events from anthropogenic maintenance activities which can reset or alter community succession. Pinn et al., (2005) found that assemblages on young structures were dominated by ephemeral species whereas older structures were colonised by perennial species which is consistent with natural shore succession (Chapman & Underwood, 1998), although with age, benthic communities on artificial structures may become more alike to natural shores (Burt et al., 2011). Biodiversity on an artificial structure is also influenced by whether or not it has been enhanced to provide suitable habitat for marine fauna and flora. Yet regardless of ecological enhancement, a crucial factor in determining the colonisation and community composition of artificial structures is the proximity to source population and adequate larval supply, which could be from either natural shores or other artificial structures (Cowen & Sponaugle, 2009). This thesis investigates the influence of substrate on existing benthic communities on wooden and rock groynes and the ecological process involved in structuring the benthic communities specifically settlement, recruitment and mortality.

1.3 Connectivity

Connectivity is defined as the potential for geographically separated populations to exchange individuals (Pineda et al., 2007). This is influenced heavily by spatial and temporal variation in transport processes and propagule supply (Cowen & Sponaugle, 2009). The proliferation of artificial structures is resulting in further habitat fragmentation within the marine environment and understanding the connectivity between habitats is vital for management and conservation (Bishop et al., 2017).

Artificial structures built in soft sediment environments can create ‘islands’ of hard substrate which can support species previously absent due to the lack of suitable habitat. Island biogeography theory states that the number of species found on an island is dependent on the colonisation and extinction rates of species (MacArthur & Wilson, 1967) which will vary depending on the size of habitat and distance from source populations. The larger the artificial structure the greater potential to support a greater abundance and species richness of organisms, as smaller islands will have a lower carrying capacity resulting in higher rates of extinction (MacArthur & Wilson, 1967). The proximity to source populations is key for successful recruitment of species and the position of a species within their metapopulation will determine its ability to colonise an unoccupied habitat (Levins, 1969). Metapopulation theory emphasises the connectivity between spatially separated subpopulations to provide long term survival of the overall metapopulation (Levins, 1969). Each subpopulation has varied rates of disturbance which can alter community composition and can be tested using patch dynamic theory (Pickett & White, 1985; Raffaelli & Hawkins, 1996). If a population on a small isolated ‘island’ were to suffer a large disturbance in which the majority of the population were destroyed, the re-colonisation of this habitat would be reliant on the connectivity between adjacent habitats. These ‘islands’ of artificial habitat have the potential to exchange larvae and/or individuals with natural rocky shores and other artificial structures if suitable conditions are met, allowing the exchange of genetic diversity. The increased addition of artificial structures also increases the provision of ‘stepping stones’ for the dispersal (Kimura & Weiss, 1964) and range extensions of both native (Johannesson & Warmoes, 1990) and non-native species (Bulleri & Airoldi, 2005). The intertidal snail *Littorina saxatilis* is a direct developer and has limited dispersal ability, yet rapidly colonised breakwaters on the Belgian coast which, provided stepping stones for dispersal on an otherwise sandy shore (Johannesson & Warmoes, 1990). For sessile benthic species the stepping stone theory is highly influenced by supply side ecology, in which larval supply is determined by reproductive success and physical constraints such as food availability, temperature and hydrodynamic processes (Kimura & Weiss, 1964; Lewin, 1986). This thesis investigates the potential for groynes and piers to become ‘islands’ of fragmented habitat which can function within the metapopulations of the region.

1.3.1 Non-native species

The processes by which artificial structures can act as stepping stones for improving and facilitating the dispersal of marine organisms has recently been studied in more detail specifically for non-native species and climate migrants (Bulleri et al., 2006; Keith et al., 2011; Airoldi et al., 2015). Non-native species are known to be more prevalent on artificial structures in comparison to nearby natural rocky shores (Mineur et al., 2012). They can be transported across major geographic barriers by human activities which can range from shipping ballast water to unintentional escapes (Ruiz et al., 1997). The construction of artificial structures increases the opportunities for non-native species to establish due to the provision of new bare substrate (Lambert & Lambert, 2003; Simkanin et al., 2012). For example, pontoons and pilings have created available habitat for the spread of non-native invasions, particularly in estuaries (Glasby et al., 2007). Due to opportunistic behaviours and quick reproductive strategies non-native species are able to rapidly colonise freshly constructed and or disturbed surfaces (Airoldi & Bulleri, 2011; Airoldi et al., 2015).

Non-native species are known to have large impacts on natural marine ecosystems including altered competition for food and space, habitat modification and the introduction of new pathogens and exotic genes (Streftaris et al., 2005) , the majority of these impacts are irreversible. The green seaweed *Codium fragile* is native to East Asia but has been introduced into the Adriatic Sea where it is spreading rapidly across coastal defence structures, and on breakwaters it has shown to establish a viable population and monopolise space in the sheltered lower shore habitats (Bulleri & Airoldi, 2005).

Non-native species are known to be associated with artificial structures for at least part of their life-cycle (Ruiz et al., 2009) and it is expected that the continued addition of artificial structures will subsequently lead to the increased spread of non-native species, especially in areas where hard substrate would otherwise be absent. This thesis investigates the connectivity and potential stepping stones between fragmented patches of natural and artificial habitat through the use of hydrodynamic modelling.

1.3.2 Climate migrants

Climate change poses a worldwide threat to biodiversity which is set to increase over time (Rosenzweig et al., 2007; IPCC, 2014). Climate change includes increased air temperatures, increased sea temperatures and increased sea level rise. The upper 75m of the global oceans have been warming at an average rate of 0.11^{o}c per decade between 1971 and 2010 (Rhein et al., 2013). Climate migrants are species which are responding to changing meteorological and oceanographic variables such as air and sea temperatures by shifting their geographic ranges. The construction of hard artificial substrate in the marine environment provides species with novel habitat for colonisation from which geographical barriers could potentially be crossed (Bulleri & Airoldi, 2005; Glasby et al., 2007; Adams et al., 2014). Species may be directly or indirectly affected by climate change for example sea surface temperature (SST) can directly impact on reproductive strategies of marine organisms whereas the arrival of habitat-forming species, such as coral, can indirectly effect the survival of existing species due to reduced available space for colonisation (Yamano et al., 2011).

Range expansions can be a response to direct physical variables such has temperature increases (Herbert et al., 2003; Chen et al., 2011; Rius et al., 2014) or indirect through the arrival of habitat forming species which can facilitate colonisation of subsequent species (Yamano et al., 2011). Range extensions start with the initial arrival of a few individuals, which if they survive is followed by population growth and the establishment of a persistent population (Bates et al., 2014). In contrast, range reductions occur gradually commencing with a decline in performance leading to a population decrease and finally result in a local extinction (Bates et al., 2014). Range reductions can be the consequence of population decline through temperature increases (Jones et al., 2010) or indirectly through the decline of food availability and suitable habitat. Temperature driven range extensions have already been reported for various marine species including seaweed (Smale & Wernberg, 2013), invertebrates (Mieszkowska et al., 2005) and fish (Perry et al., 2005) and further temperature rise could result in widespread extinctions and altered communities due to changes in competition between new and existing species (Perry et al., 2005). Cold-water adapted species are reducing in abundance and retreating pole wards while warm-water species are increasing in abundance and advancing (Hawkins et al., 2008). In the UK, significant eastward range expansion of southern intertidal invertebrate species have been recorded, including *Perforatus perforatus, Gibbula umbilicalis* and *Patella depressa* (Mieszkowska et al., 2005). The addition of artificial structures may act as “stepping stones” to increase the dispersal potential for climate migrants (Adams et al., 2014; Airoldi et al., 2015). Moschella et al (2005) & Keith et al (2011) revealed range shifts due to climate change are causing species to shift spatially (generally northwards), with artificial structures often being used as refuges for these species.

1.3.3 Mobile populations

A significant body of research has been conducted on the mobile fauna including fish and crustaceans associated with artificial reefs (Ambrose & Swarbrick, 1989; Ambrose & Anderson, 1990; Bohnsack et al., 1991; Jensen et al., 1993, 2000; Baine, 2001; Perkol-Finkel et al., 2006), yet coastal structures such as piers and breakwaters have been neglected. The influence piers have on surrounding mobile communities has rarely been investigated in the marine environment and most existing studies have been conducted in estuarine habitats (Glasby, 1999a; Metzger et al., 2001; Coleman & Connell, 2006; Able et al., 2013). Larger artificial structures such as piers and jetties which extend into the subtidal zone have the potential to support fish populations through the provision of shelter, foraging areas and/or nursery habitat. Fish are known to aggregate around changes in topography (Randall, 1963) therefore in soft sediment environments, coastal artificial structures may become fish aggregating devices (FAD) as found with offshore wind farms (Wilhelmsson et al., 2006). Research comparing artificial structures with natural reefs has found that species assemblages differ and highlights an uncertainty as to whether or not artificial structures can support viable populations of fish (Clynick et al., 2007, 2008). To date no detailed research has been conducted on the food chains around piers and other coastal infrastructure.

Clynick et al., (2007) discovered that the amount of epifauna on pier pilings is positively correlated to the abundance and diversity of fish in Sydney Harbour, which supports similar studies conducted on natural reefs (Levin & Hay, 2002). Keough & Butler, (1979) found that asteroid predators in South Australia reduced the number of sponges and ascidians on pier pilings. However to date, no studies have investigated if the epifauna on the pier pilings provides a food resource for primary and secondary consumers in the overall food web. This thesis will investigate the distribution of mobile fauna surrounding the piers and secondly to determine if the epifauna on the pier creates a resource in the overall food web.

1.4 Ecological Enhancement

Ecological enhancement, which is also known as ecological engineering, is an emerging field of research which focuses on implementing ecologically sensitive designs into hard artificial structures to increase and/or improve the habitat for biodiversity (ITRC, 2004). These techniques can be used either at the time of construction or retrospectively on existing structures. Ecological enhancement techniques are used to improve the habitat provided and encourage colonisation without negatively impacting on the structural integrity of the asset.

Existing trials have been conducted at various scales from variation in concrete texture (<1cm) (Coombes et al., 2015) to increased surface complexity in the form of large scale ecologically enhanced concrete panels (Perkol-finkel et al., 2017) and pre-cast habitat enhancement units (1m+) (Firth et al., 2014b), see Table 1.1 for a detailed list. Experiments have also been conducted into the types of concrete used and alternative concrete mixes which could promote biodiversity (Sella & Perkol-Finkel, 2015; Dennis et al., 2017). To date, successful ecological enhancement techniques have increased the microhabitats on artificial structures through improved surface heterogeneity and areas of water retention (Firth et al., 2014b). Increased surface heterogeneity can be achieved by drilling various sized pits into seawalls and/or boulders (Martins et al., 2010; Firth et al., 2014b) or attaching precast concrete tiles to surfaces (Borsje et al., 2011; Loke et al., 2015). Water retention can be achieved using a variety of techniques including drilling large cores into boulders (Evans et al., 2015) or incorporating moulded concrete voids orientated to retain water into concrete structures (Firth et al., 2016b). On larger scales, pre-cast habitat enhancement units can be deployed, which have shown to increase the diversity of species within artificial coastal structures (Firth et al., 2014b; Perkol-finkel et al., 2017). This thesis will investigate the use of low cost ecological enhancement techniques which can be used on varying rock armour structures together with monitoring the use of retrofitted artificial rock pools on seawalls within temperate climates.

Table 1.1: Existing ecological enhancement trials on artificial structures in intertidal habitats (See Firth et al., (2016b) for a comprehensive list).

Treatment	Location	Findings	References
Pits and Grooves			
Drilled pits into Basalt seawalls (10-12mm deep).	Azores, Portugal	Increased number of limpets in areas with pits compared to bare seawall.	(Martins et al., 2010)
Various sized pits and groves in sandstone seawall blocks	Sydney Harbour, Australia	Smaller pits supported high numbers of limpets and chitons.	(Chapman & Underwood, 2011)
Pits in concrete breakwater (14mm and 22mm diameter)	Plymouth, UK	Increased number of species in the treatments. No difference between pit sizes.	(Firth et al., 2014b)
Manipulating wet mortar in concrete seawall clad with stone (pits, recessed crevices and grooves)	Shaldon, UK	Increased number of species in the recessed crevices and pits.	(Naylor et al., 2011; Firth et al., 2014b)
Concrete tiles and slabs			
Concrete panels of varying complexity (30cm x 30cm) on low-crested structures.	Elmer, UK	At both <1cm and <10cm scales the increased surface complexity resulted in an increased species diversity.	(Moschella et al., 2005)
Concrete tiles containing varying percentages of crustose coralline algae covered coral rubble (CCACR)	Singapore	Preferential settlement of the fluted giant clam *Tridacna squamosal* on the higher percentage of CCACR.	(Neo et al., 2009)
Concrete slabs (75cm x 50cm) with grooves and pits attached to rock armour	Netherlands	Lower slabs supported higher number of species. Mussels developed best in grooves.	(Borsje et al., 2011)
Concrete tiles with fine scale surface textures	South west England, UK	Grooved tiles had a larger abundance of barnacles than smooth concrete.	(Coombes et al., 2015)
Concrete tiles of varying complexity fixed to rock armour	Singapore	Greater species richness recorded on tiles with highest complexity.	(Loke et al., 2014, 2015; Loke & Todd, 2016)
3D printed concrete tiles with varying complexities	Sydney Harbour, Australia	Variation between sites. One site showed a higher species richness on the more complex tiles.	(Strain et al., 2017)
3D Printed tiles with varying designs attached to rock groynes	Highcliffe, UK	Good early successional colonisation. Severe loss of tiles.	(Sherrard, 2017)
Bio-enhanced	Israel	Increase in live cover, richness	(Perkol-Finkel et

ECOncrete® panels on seawall in a marina		and biodiversity. Mobile fauna particularly enhanced by holes and crevices.	al., 2017)
Water retaining techniques			
Omitting blocks and creating a cavity with a lip	Australia	Increased diversity especially on the high shore.	(Chapman & Blockley, 2009)
Flowerpots attached to seawalls.	Sydney Harbour, Australia	Increased number of species within the flowerpots compared to the seawall at both high and midshore tidal heights. Shallow mid shore pots had highest abundance and diversity.	(Browne & Chapman, 2014)
Drill-cored rock pools on a granite breakwater (15cm diameter, 12cm & 5cm deep)	Tywyn Wales, UK	Pools supported a greater number of species than breakwater. Different community structure compared to natural pools	(Evans et al., 2015)
Concrete poured rock pools in SHED units (~1250cm^3) at two tidal heights on two exposures.	Galway Bay, Ireland	The lower pools supported a higher number of taxon than the upper pools on the exposed side only.	(Firth et al., 2016b)
Artificial turf (coir) used inside flowerpots attached to seawalls to attract native species.	Sydney Harbour, Australia	Native species occupied 95% of the space within flowerpots. Non-native species decreased over time.	(Morris et al., 2017)
Pre-cast habitat enhancement units			
BIOBLOCK (1.5m x 1.5m x 1.1m), habitat enhancement unit includes rock pools, pits, crevices.	Colwyn Bay, Wales, UK	The variety of habitats within the BIOBLOCK supported a higher number of species than adjacent boulders.	(Firth et al., 2014b)
Econcrete® rock pools installed within rock armour	USA	Rock pools supported a range of natural rock pool species.	(Firth et al., 2014b)
Large concrete face panels for seawalls, with textures including steps and cobblestones	Seattle, USA	Results pending.	Cordell et al (In press)

1.4.1 Policy Drivers

In Europe there are various policies which can be used to implement ecological enhancement (Table 1.2). The EU Water Framework Directive (Directive 2000/60/EC) is one of most influential as it provides targets in which all heavily modified waterbodies (HMWB) should aim to achieve. HMWB include areas such as ports, harbours and armoured coastlines. The Environment Agency has produced a mitigation measures manual (Environment Agency, 2013) which can be used to achieve good ecological status including the use of ecological enhancements. The European Marine Strategy Framework Directive (MSFD) contains other influential policies which mitigate against the degradation of habitat through "corrective measures" which supports the use of ecological enhancement techniques.

Within the UK there is very little legislation which enforces engineers and construction companies to use ecologically sensitive designs within their structures. However the UK's Marine Policy Statement (UK Parliament, 2011) recommends that in addition to avoiding harm to marine biodiversity developers should also "where appropriate provide opportunities for building-in beneficial features". This promotes the use of ecological sensitive designs and ecological enhancements in new developments. Details of other UK policies which can be used to support ecological engineering can be seen in Table 1.2. It has however been noted that improvements to legislation are required in order to improve the ecologically sensitive design of structures and create more opportunities for ecological enhancement (Naylor et al., 2012; Evans et al., 2017).

Table 1.2: European and UK Legislation relating to the protection and enhancement for coastal areas which could be used to facilitate ecological enhancements (adapted from Naylor et al., 2011, 2012).

Level	Legal framework	Details
European		
	EC Water Framework Directive	Requires that all waterbodies including ports, harbours and defended coastlines achieve "good ecological status".
	Habitats Directive (92/43/EEC)	Emphasises the importance of ecological connectivity between habitats and species. Allows the creation of Special Areas of Conservation (SAC).
	Birds Directive (79/409/EEC)	Creation of Special Protection Areas (SPAs)
	Strategic Environmental Assessment (SEA) Directive (2001/42/EC)	The Directive clearly provides opportunities for consideration of measures to enhance as well as mitigate against significant impacts on the environment
	Marine Strategy Framework Directive (2008/56/EC)	To achieve 'Good Environmental Status' in European seas by 2020
	Environmental Impact Assessment (EIA) directive	Aimed to assess all environmental consequences of development and provide mitigation measures if needed.
UK		
	Marine and Coastal Access Act 2009	Enhance the marine environmental for current and future generations. Creation of Marine Conservation Zones (MCZ)
	Planning Policy Statement 9 (Biodiversity and Geological Conservation, 2005)	Provides a requirement to incorporate biodiversity enhancement into planning policies and planning decisions.
	Biodiversity: The UK Action Plan (1994)	Improving the extent and abundance of priority habitats and species.
	Environment Act (1995)	The duty to 'conserve' could include compensation to developments adversely impacting waterbodies
	Harbour Revision Orders (Harbour Act, 1964)	Ecological enhancements may be required as part of these permissions.
	Countryside and Rights of Way Act 2000	Supports habitat protection and enhancement, and places a requirement on local authorities to have regard for biological conservation and enhancement in planning

1.5 Thesis Structure

This thesis addresses six knowledge gaps in existing literature, the outcome of which could be used to inform the construction and ecological enhancement of new and existing artificial structures. Each chapter will focus on a particular aspect of artificial structure ecology and the processes involved in structuring the communities on and around them. The overall aim of the thesis is to quantify the variation in biodiversity on different artificial coastal structures compared to natural habitats and assess the magnitude of species interactions, connectivity and the potential for ecological enhancement.

Chapter 1: General Introduction

Provides an overview of existing knowledge regarding the ecology and ecological enhancement of artificial coastal structures including information on non-native species, climate migrants and mobile populations. This chapter provides the background information for Chapters 2-7 and highlights the importance of studying artificial structures.

Chapter 2: The Benthic communities Associated with Wooden and Rock Groynes in Comparison to Natural Rocky Shores

This chapter investigates the existing benthic communities on wooden and rock groynes along the south coast of England with a focus on the influence of substrate type on community composition. Groynes are common coastal defence structures which are constructed to maintain beach levels and prevent coastal erosion, yet there has been very little research on their associated benthic ecology. Understanding the variation in communities present on wooden and rock groynes will allow engineers and researchers to determine the potential spread of non-native species, particularly around ports and harbours which are known introduction points for non-natives.

Chapter 3: The Recruitment and Mortality of Key Intertidal Species on Wooden and Rock Groynes

This chapter builds on the previous chapter and investigates the ecological processes involved in structuring the benthic communities associated with artificial structures, specifically settlement, recruitment and mortality. The focus is on the influence of substrate on settlement and recruitment of key species on both natural rocky shores and artificial structures. Many artificial structures are isolated; therefore it is important to understand the persistence of populations within assemblages and their potential contribution to coastal ecosystems on local and regional scales.

Chapter 4: The Influence of Piers on Surrounding Communities of Mobile Fauna

Very little research has been conducted on the influence coastal artificial structures can have on the surrounding mobile fauna, particularly in terms of their contribution to the food web dynamics. This chapter investigates the influence artificial structures such as piers can have on the surrounding communities and food webs through the use of Baited Remote Underwater Video (BRUV) and Stable Isotope Analysis (SIA). BRUV surveys were used to quantify the distribution and diversity of mobile fauna around the pier structures. Understanding how structures such as piers influence the mobile communities is important in order to appreciate the impact of future installations on the overall community dynamics of the region.

Chapter 5: Do Inshore Structures act as Stepping Stones for Dispersing Native and Non-native Species?

Understanding the connectivity between fragmented patches of habitats is important in order to conserve and manage marine landscapes. This chapter uses a coupled bio-physical model to determine the connectivity between natural and artificial structures and determines the potential for a series of artificial structures to become stepping stones for dispersal. The case study is centred in Poole Bay which encompasses Poole Harbour to the west, a known entry point for invasive species and the bay itself which consists of a series of 80+ intertidal groynes located within a 16 km stretch of soft sediment coastline. The region is also known to be at the edge of local biogeographic ranges specifically for climate migrants. The models will test the connectivity between natural and artificial habitats and also the potential for artificial habitats to become source populations for subsequent dispersal.

Chapter 6: Evaluation of Low-Cost Ecological Enhancement Techniques to Improve Habitat Heterogeneity on Coastal Defence Structures

This chapter evaluates the application of low cost ecological enhancement techniques on coastal defence structures in locations exposed to high wave energy. Inspired by holes and groove ‘blast features’ caused by quarrying, the aim was to create designs that could be replicated on any boulder defence structure, both during construction and retrospectively. This study was conducted on both granite and limestone boulders to compare the application on two substrate types, which has previously never been investigated. Habitat heterogeneity is known to affect the species richness and diversity of communities and increased heterogeneity provides refuge from both environmental conditions and predators. Refugia from environmental conditions and predation is a feature which is commonly absent from coastal defence structures, this chapter aims to investigate the impact of creation of refugia in an attempt to increase the biodiversity of rock armour.

Chapter 7: Shelving the Coast with Vertipools: Retrofitting Artificial Rock Pools on Coastal Structures as Mitigation for Coastal Squeeze

Sea level rise is resulting in coastal squeeze, which is creating a compressed intertidal zone and resulting in loss of habitat. This chapter evaluates the novel use of retrofitted artificial rock pools on seawalls as mitigation against the impacts of coastal squeeze. Seawall communities are known to be less diverse than natural shores due to their tidal height and intense wave exposure. Water retention has been identified as limiting biodiversity of artificial structures, therefore creating water retaining features such as artificial rock pools could enable more species to survive on the seawalls. Through the involvement of artists and engagement of school children multifunctional artificial rock pools were designed, constructed and monitored.

Chapter 8: Discussion

This chapter summarises the findings of the research and discusses the implications of the results in the context of management and conservation. The key findings are highlighted in the context of existing research and reinforcing the importance of the studies. The limitations of the research are identified and future studies are suggested in the light of knowledge gaps.

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2 The Benthic Communities Associated with Wooden and Rock Groynes in Comparison to Natural Rocky Shores

Abstract

A comparative study of intertidal benthic communities associated with wooden and rock groynes and natural shores was conducted on the south coast of England on a local and regional scale. Although the natural shores were relatively species rich and diverse compared to the artificial substrates, wooden groynes had higher mean species richness and species diversity compared to the rock groynes. Wooden groynes were also found to have the highest community similarity to natural shores, which is partly due to their higher cover of fucoid algae. The natural shores had greater abundances of mobile grazers, including the gastropods *Patella vulgata, Littorina littorea* and *Gibbula umbilicalis,* whereas the groynes were dominated by sessile species, such as the green alga *Ulva* spp. and barnacle *Semibalanus balanoides.* Overall, there were significant differences in assemblages between the wooden and rock groynes, and at regional scales between the artificial structures and natural shores at mid-tidal levels. On local and regional scales there was very high spatial variability of communities on these artificial structures due to differences in structure, design, age, maintenance, scouring and tidal height. These different assemblages may interact with populations on natural intertidal reef habitats in these regions including protected areas, with implications for species interactions and community dynamics.

2.1 Introduction

Due to rising sea levels and predicted higher storm frequency, coastal defence structures are increasing worldwide (Pethick, 2001; Wang et al., 2012). Currently, 46% of the British coastline is protected by artificial beaches and/ or structures (Masselink & Russell, 2010), including seawalls, breakwaters and groynes. Whilst artificial structures may act as surrogate habitats for rocky shores (Southward & Orton, 1954; Thompson et al., 2002; Martin et al., 2005), they generally support less diverse assemblages (Moschella et al., 2005; Mineur et al., 2012) with higher numbers of non-native species (Vaselli et al., 2008), but lower abundances of mobile fauna (Chapman, 2003; Firth et al., 2016b). Due to the dynamic environment, marine artificial structures are prone to regular disturbance events through scouring, abrasion and maintenance (Airoldi et al., 2005; Airoldi & Bulleri, 2011; Firth et al., 2014a).

Groynes are built to prevent coastal erosion and are a feature of sedimentary shorelines globally (Figure 2.1). Wooden groynes create a solid barrier to sediment movement by intercepting longshore drift, whereas rock groynes are designed to both dissipate wave energy and to trap and retain sediment (Dong, 2004). They are constructed perpendicular to the shoreline (Masselink & Hughes, 2003) and are constructed of wood, rock or concrete, which have different life expectancies and functions (French, 2001). Wooden groynes in Britain are predominately constructed from “greenheart” *(Chlorocardium rodiei)* or “Ekki” *(Lophira alata),* imported for its strength and longevity, and are intended to last for approximately 25 years, with periodic maintenance required to ensure structural integrity (French, 2001). Rock groynes usually consist of multiple boulders (1 to 8 tonnes) placed in piles perpendicular to the shore, extending to Mean Low Water mark and are predicted to survive at least 50 years with minimal maintenance (Dong, 2004). Wooden groynes, being narrower than rock groynes have a smaller footprint on the seabed, resulting in less space taken up on the shore by the structures. For this reason, they tend to be more popular on crowded tourist beaches (Dave Harlow, pers. comm).

Figure 2.1 Groynes sampled along the south coast of England a) wooden groyne with fucoid canopy cover in Region 2, b) damaged wooden groyne with planks missing in Region 1, c) rock groyne toe in Region 1.

The construction of groynes on soft sediment shores can provide the dominant or only hard substrata in an area, thus providing novel habitat for colonisation of species previously absent (Johannesson & Warmoes, 1990; Airoldi et al., 2005). As substrate type, surface texture and rock mineralogy have been shown to influence recruitment and assemblage composition (Caffey, 1982; Ambrose & Swarbrick, 1989; Holmes et al., 1997; Herbert & Hawkins, 2006; Green et al., 2012; Liversage et al., 2014), the materials used to construct groynes could determine the local patterns of species diversity (Firth et al., 2015). In addition, previous work in southern England suggested species richness increases with the age of rock groynes (Pinn et al. 2005).

Due to the design and orientation of groynes, habitats with varied wave exposure, temperature and shading are created on windward and leeward sides. Variation in temperature and shading has shown to have significant effects on the communities present on natural rocky shores (Kensler, 1967; Little et al., 2009; Miller et al., 2009) and existing artificial structures (Blockley & Chapman, 2006). Algae has been shown to be more abundant in unshaded areas than shaded areas (Glasby, 1999b; Clark et al., 2004), whereas benthic sessile invertebrates have been recorded in greater numbers in shaded areas due to the reduced thermal stress (Bertness et al., 1999; Glasby, 1999b; Blockley & Chapman, 2008). These habitats with varied temperature/shading may create refuges for climate migrants, including barnacle *Chthamalus* spp., limpet *Patella depressa* (Mieszkowska et al., 2005; Hawkins et al., 2008) and the gastropod *Gibbula umbilicalis* (Keith et al., 2011), as well as non-native species (Bulleri et al., 2006; Mineur et al., 2012), and in turn alter the assemblage composition (Connell & Glasby, 1999; Glasby & Connell, 2001).

Structures built at different tidal heights can also affect the local vertical distribution of species (Pinn et al., 2005) due to variation in temperature fluctuations and submersion time (Raffaelli & Hawkins, 1996). Artificial structures such as groynes which extend from the HWM to ELWS provide a variety of habitat conditions for organisms. The landward end of the groyne will experience very dry warm conditions whereas the seaward end of the groyne will experience periods of submersion at high tide and exposure at low tide.

The aim of the research was to thus compare the diversity and composition of benthic communities on wooden and rock groynes with natural rock substratum. The following hypotheses were tested:

(1) Wooden and rock groynes have a lower species richness, total abundance and species diversity compared to the natural substratum.

(2) Rock groynes have a higher species richness, total abundance and diversity compared to the wooden groynes, as rocky shore communities are more similar to those natural rocky shores;

(3) Variation in the assemblages between natural and artificial structures at different scales can be explained by differences in substrate type, aspect, distance down the shore, and tidal zone.

2.2 Methods

2.2.1 Study Sites

Four coastal cell regions were surveyed along the south coast of England: Region 1-Poole/Purbeck, Region 2-Solent/Isle of Wight, Region 3-Worthing and Region 4-Hastings (Figure 2.2). Within each region one shore site termed “artificial shore” with both wooden and rock groynes in close proximity (<180m) was sampled, together with the two closest and most accessible nearby natural rocky shore sites, giving three sampled shore sites per region (one artificial and two natural) (Table 2.1; Figure 2.2). Sites were chosen to ensure that wave exposure, gradient and aspect were as similar as possible. Bournemouth, Worthing and Hastings are all holiday destinations with groynes constructed to maintain beach levels and prevent sediment loss through longshore/littoral drift. Lee-on-Solent is situated at the mouth of Southampton Water where the groynes intercept sediment transportation to maintain clear channel navigation. The south coast has a south–west prevailing wind direction and all of the study sites are moderately exposed to wave energy. Generally, the residual current flow is from west to east. All sites have a shallow shore gradient and the artificial structures were surrounded by sand or shingle (Table 2.1).

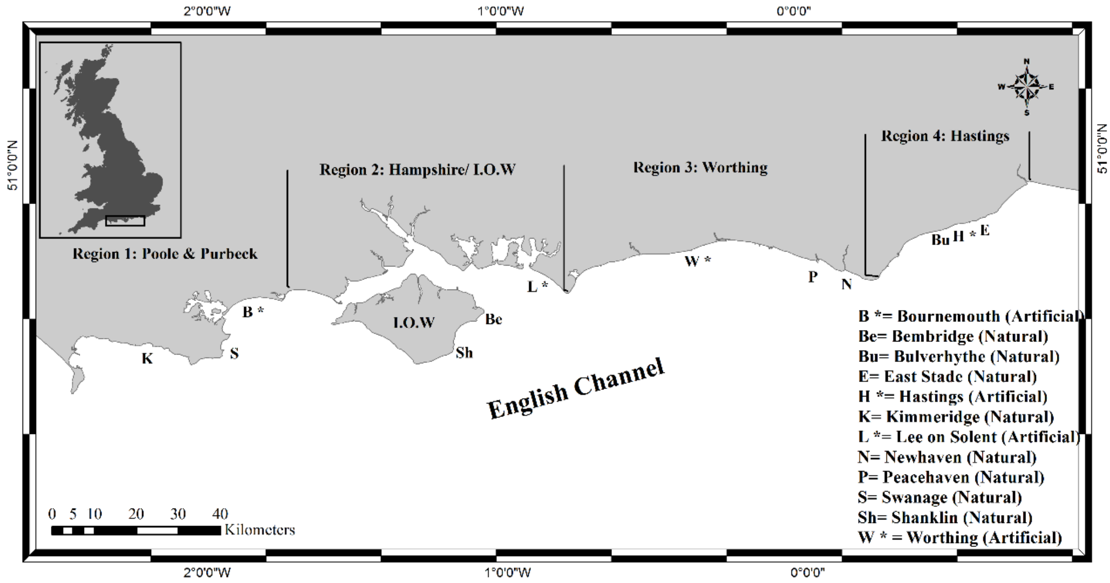

Figure 2.2 Central south coast of England showing locations of natural and artificial survey sites within each region. At each 'Artificial site' there were both wood and rock groynes. From west to east regions are Region 1- Dorset; Region 2- Hampshire and Isle of Wight; Region 3- Worthing; Region 4-Hastings (* Indicates artificial shore).

Table 2.1 Details of groynes sampled in the different regions and the comparable natural rocky shores and substrate types.

Region	Groyne Type	Construction Date	Total length of groyne	Tidal range neaps-springs (m)	Shore substrate type	Natural rocky shores (Site and type)
Region 1: Bournemouth	Wood (Greenheart)	1991	30m	1.3-1.7	Soft sediment	Swanage (north)-sandstone boulders
	Rock (Portland Limestone)	2007	30m	1.3-1.7	Soft Sediment	Kimmeridge-dolomite platforms
Region 2: Lee on Solent	Wood (Jarrah)	1971	30m	2.9-4.4	Shingle & mixed sediment	Bembridge-limestone ledges
	Rock (Portland Limestone)	1995	30m	2.9-4.4	Shingle & mixed sediment	Shanklin-sandstone ledges
Region 3: Worthing	Wood (Douglas Fir)	1989	60m	1.6-6.5	Shingle & course sediment	Peacehaven-sandstone boulders
	Rock (Limestone)	2005	60m	1.6-6.5	Shingle & course sediment	Newhaven-sandstone boulders
Region 4: Hastings	Wood (Greenheart)	1980	60m	3.2-7.2	Shingle & fine sand	Bulverhythe-sandstone boulders
	Rock (Granite)	2006	60m	3.2-7.2	Shingle & fine sand	East Stade-sandstone boulders

2.2.2 Field Sampling

All sites were surveyed between June and September in 2015 (R1, R2) and 2016 (R3, R4). Sites within each region were sampled on consecutive days to facilitate accurate positioning of sampling points on groynes and on natural shores in relation to tidal height. These were determined by using tidal curves for each site and observing the duration of exposure and submergence of the shore profile and structures. On both natural and artificial shores quadrats were sampled on vertical/sloping surfaces where possible.

Groynes

Within each artificial site (one per region), three wooden and three rock groynes with a west and east-facing aspect were chosen in close proximity (<180m), allowing for direct comparisons of existing communities between substrates. For each aspect, the length of the groyne measured and subsequently divided into three equal sections, from the Extreme High Water mark down towards Mean Low Water mark. These equal length sections are referred to as the Inner section (shore side), Central section and Outer section (seaward) respectively (Figure 2.3). Each section was then divided into two subsections by vertical tidal height, referred to as Upper and Mid Zone (Figure 2.3). The lowest sampling point was between Mean Tide Level and Mean Low water Springs Tide Level.

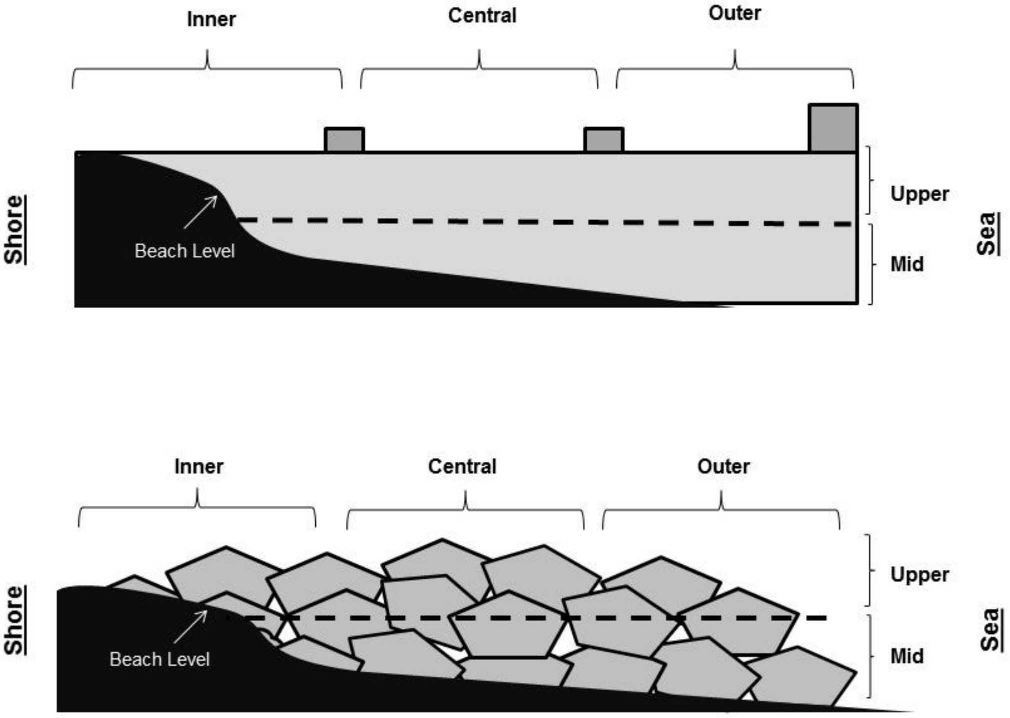

Figure 2.3 Wooden and rock groynes showing sampling sections distance: inner, central and outer and tidal zone: upper and mid.

Three 0.25 m^2 quadrats were placed haphazardly on vertical/sloping surfaces within each subsection on each groyne (Total n = 18 quadrats on each aspect of each groyne). The percentage cover of algae and mussels and count data for all other species were recorded and this protocol was used on both the west facing (prevailing up-drift/exposed) and east facing (prevailing down-drift/sheltered) sides of each groyne (Total n=36 quadrats for each groyne). Three measurements of substrate surface temperature were obtained using a Raynger ST60 temperature gun in each sub-section on both sides of the groynes.

Natural shores

Two natural rocky shores were surveyed in each region using comparable methodology. Where possible, three transects placed on the natural shores were of similar distance apart as the groynes (<180 m). Three transects matching the lengths of the groynes in each region were laid perpendicular to the shore from extreme high water mark towards mean low water. Transects were divided into three equal sections: Inner (shore side), Central section and Outer section (seaward). Where boulders were present, these sections were divided vertically according to upper and middle tidal levels. Three 0.25 m^2 quadrats were placed haphazardly on each side of the transect within each subsection (Max n = 18 for each transect if boulders were present). Due to the absence of rock pools on the groynes, these areas were avoided on the natural rocky shores. As some shores were mostly flat reefs, differences in substratum aspect could not be tested. The percentage of sand and shingle within the quadrats were also recorded and any quadrats containing 100% sediment were omitted.

2.2.3 Data and statistical analyses

The main aim was to assess biotic differences between the three types of substrate sampled, namely wooden groynes, rock groynes and natural shores. For the purpose of statistical analysis, the factor “Transect” is used to denote the different groynes on artificial sites and the individual transects on natural sites. Species richness (S), total abundance (N) and Shannon Wiener species diversity (H) of communities were calculated for each quadrat using the DIVERSE function in PRIMER-e V6 (Clarke & Gorley, 2006). Using R v3.2.4 (R core team, 2016), statistical differences in S, N, H were tested with linear mixed effect models (LMMs) to assess the differences between fixed effect factors (Substrate, Distance and Zone) and the variance between random effect factors (Region, Site, Transect) (Long & Ervin, 2000). Region was included as a factor as they represent sediment transport cells which have the potential to be biologically different due to larval supply (Motyka & Brampton, 1993; Herbert et al., 2007). The factor Site is nested within Region and the factor Transect is nested within Site. Model selection was determined by backwards stepwise selection and the Akaike Information Criterion (AIC) was used to evaluate the significance of the variables in the various model. Histograms were plotted to visually assess and confirm that the residuals were normally distributed (Zuur et al., 2009) and transformation of taxonomic data were used where necessary to improve residual normality.

To solely compare the communities on wooden and rock groynes, all the data from natural shores in each region were excluded and the LMM were run using R v3.2.4 (R core team, 2016) to test the statistical differences in S, N, H with Substrate (wooden and rock), Aspect (east/ west), Distance (Inner, Central, Outer) and Zone (Upper, Mid). A Canonical Analysis of Principal coordinate (CAP) and SIMPER analysis were then used to identify the species contributing to the difference in community similarity on wooden and rock groynes.

2.3 Results

2.3.1 Natural versus Artificial: Substrate

A total of 11 species were recorded across both the upper and mid zones on the natural shores compared to only 8 that were recorded on the artificial structures. The number of species varied by region, with Region 2 having the greatest number of species on artificial shores (S=8) and Region 3 having the lowest (S=2). The species richness on natural shores was the same for Region 1, 3 and 4 (S=11), however, the natural shores in Region 2 only had a recorded species richness of 9.

The first fitted models allowed for the fixed effects of substrate type, distance and zone within each transect or groyne (collectively denoted TG), within each site, within each region (Table 2.2). Differences between the three substrate types were broken down into two independent one degree of freedom contrasts (i) natural shores vs all artificial groynes and (ii) wooden groynes vs rock groynes. There was a significant difference in the species richness, total abundance and species diversity between substrates, although the significant difference was between wooden and rock groynes rather than natural and artificial shores (Table 2.2). There was no significant difference in species richness, total abundance and species diversity between natural and artificial shores (Table 2.2, Figure 2.4), although there was significant difference between natural and artificial substrates in community composition (ANOSIM, Global R= 0.061, P<0.01), with the natural shore communities having a greater abundance of mobile fauna, including the gastropods *Littorina littorea and Gibbula umbilicalis*. However, the ANOSIM revealed significant differences in communities between all three substrates (Global R=0.097, P<0.001 in all cases). The SIMPER analysis revealed high community dissimilarity between natural and artificial substrates (Table 2.3), with 9 species accounting for 91% of the overall 88% dissimilarity. When artificial substrates were separated the greatest community dissimilarity found between the natural shore and rock groyne communities (83.51%) and the lowest dissimilarity observed between the wooden and rock groynes (81.34%).

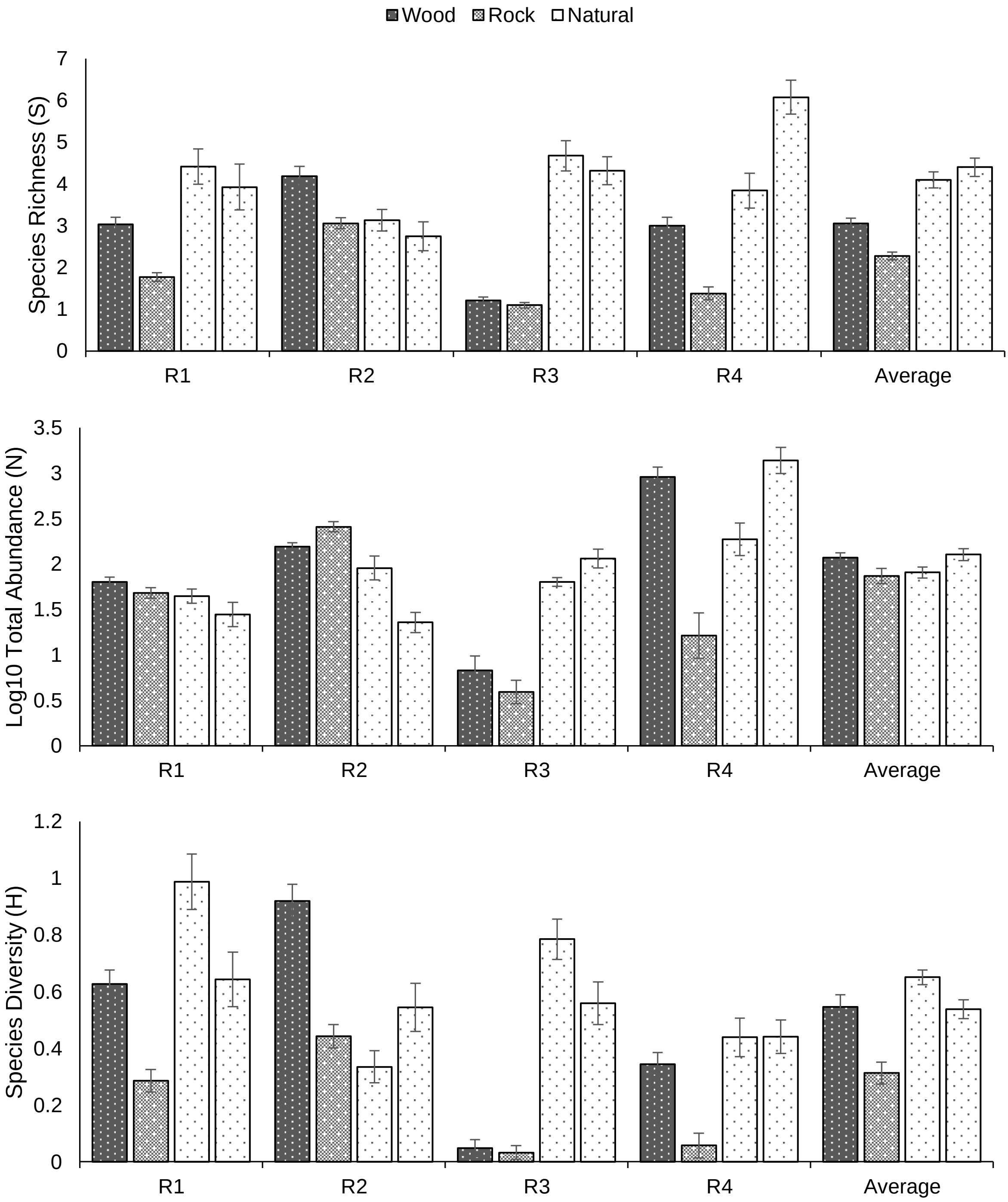

Figure 2.4 a) Mean species richness (S), b) mean Log10 total abundance (N)), c) mean species diversity (H) on wooden and rock groynes in comparison to the two natural shore sites in each region (R1-Region 1, R2-Region 2, R3- Region 3, R4-Region 4) and an average across all regions (+/- SE).

Table 2.2 Linear mixed models for a) species richness, b) total abundance, c) species diversity in relation to environmental variables- Substrate, Distance and Zone, allowing for random variation between transects/groynes (TG), sites and regions. (***=P<0.001, **=P<0.01,*=0.05 NS= Not significant).

a) Species richness (S)

Model sqrt(S)~Substrate+Distance+Zone +(1|Region/Site/TG)

Fixed Effects	**Estimate**	**Standard Error**	**t**	**P**
Intercept	1.863	0.107	17.434	***
Natural vs. Artificial	-0.128	0.061	-2.092	NS
Wooden vs. Rock groynes	0.170	0.035	4.827	***
Distance Central vs Outer	-0.176	0.035	-4.999	***
Distance Inner vs Outer	-0.426	0.066	-6.475	***
Zone Upper vs Mid	-0.301	0.040	-7.503	***

Random Effects	**Variance**	**Standard deviance**
Transect (Site: Region)	0.157	0.125
Site: Region	0.083	0.279
Region	0.000	0.000

b) Total abundance (N)

Model LogeN~Substrate+Distance+Zone +(1|Region/Site/ TG)

Source	**Estimate**	**Standard Error**	**t**	**P**
Intercept	4.640	0.571	8.122	***
Natural vs. Artificial	-0.099	0.261	-0.381	NS
Wooden vs. Rock groynes	0.461	0.184	2.511	*
Distance Central vs Outer	-0.251	0.117	-2.151	*
Distance Inner vs Outer	-0.697	0.220	-3.170	**
Zone Upper vs Mid	-0.656	0.133	-4.912	**

Random Effects	**Variance**	**Standard deviance**
Transect (Site: Region)	0.624	0.790
Site: Region	1.454	1.206
Region	0.488	0.698

c) Species diversity (H)

Model H~Substrate+Distance+Zone +(1|Region/Site/ TG)

Fixed Effects	**Estimate**	**Standard Error**	**t**	**P**
Intercept	0.587	0.089	6.534	***
Natural vs. Artificial	-0.061	0.051	-1.200	NS
Wooden vs. Rock groynes	0.167	0.027	5.997	***
Distance Central vs Outer	-0.143	0.032	-4.451	***
Distance Inner vs Outer	-0.128	0.059	-2.141	*
Zone Upper vs Mid	-0.210	0.036	-5.747	***

Random Effects	**Variance**	**Standard deviance**
Transect (Site: Region)	0.007	0.085
Site: Region	0.058	0.241
Region	0.000	0.020

Table 2.3 SIMPER table indicating average abundance of species per $0.25m^2$ quadrat on Natural and Artificial (wooden and rock groynes) shores (Ave=Mean, Cum %= Cumulative percentage contribution to dissimilarity,).

SIMPER: Artificial vs Natural			**Ave dissimilarity= 88.68**		
Species	Artificial Ave Abundance	Natural Ave Abundance	Ave Dissimilarity	Contribution %	Cum %
Semibalanus balanoides	303.15	592.37	38.64	43.57	43.57
Ulva spp.	21.44	14.50	13.52	15.24	58.81
Fucus spiralis	11.05	7.72	7.20	8.12	66.94
Mytilus edulis	19.57	210.21	5.87	6.62	73.56
Patella vulgata	8.40	2.35	4.59	5.18	78.74
Austrominius modestus	78.68	11.44	4.55	5.13	83.87
Fucus vesiculosus	0.00	4.30	2.55	2.88	86.75
Melarhaphe neritoides	3.31	0.68	2.44	2.75	89.50
Littorina littorea	0.41	2.87	1.34	1.52	91.01

2.3.2 Natural vs. Artificial: Distance

Distance down the shore had a significant influence on the species richness and species diversity of the natural and artificial shores (Table 2.2, Figure 2.5). The outer section had the greatest species richness, total abundance and diversity compared to the central and inner sections with the exception of the rock groynes in which the inner samples had the highest values (Figure 2.5).

2.3.3 Natural vs. Artificial: Tidal Zone

Tidal zone had a significant influence on the species richness, total abundance and species diversity across natural and artificial substrates (Table 2.2), with the upper zones had a significantly lower species richness, total abundance and species diversity (Table 2.2, Figure 2.6). Comparison of equivalent tidal zones also revealed a significant difference between the communities on natural shores and artificial structures (wooden and rock groynes combined) (ANOSIM, Global R = 0.093, P = 0.001). Pairwise comparisons of the tidal zones revealed that only mid-height communities were significantly different, with natural shores having community similarity than the artificial substrates (P = 0.001). SIMPER analysis revealed that 90% of the overall 82% dissimilarity between communities on the mid tidal height of natural and artificial shores was due to 14 species. The artificial sites had higher abundances of *Ulva* spp., *P. vulgata*, *Austrominius modestus*, *Idotea granulosa* and *Porphyra* sp., whereas the natural shores had higher abundances of *S. balanoides*, *M. edulis*, *F. spiralis*, *F. vesiculosus*, *L. littorea*, *G. umbilicalis, F. serratus*, *Lithothamnion* sp. and *Cladophora rupestris*.

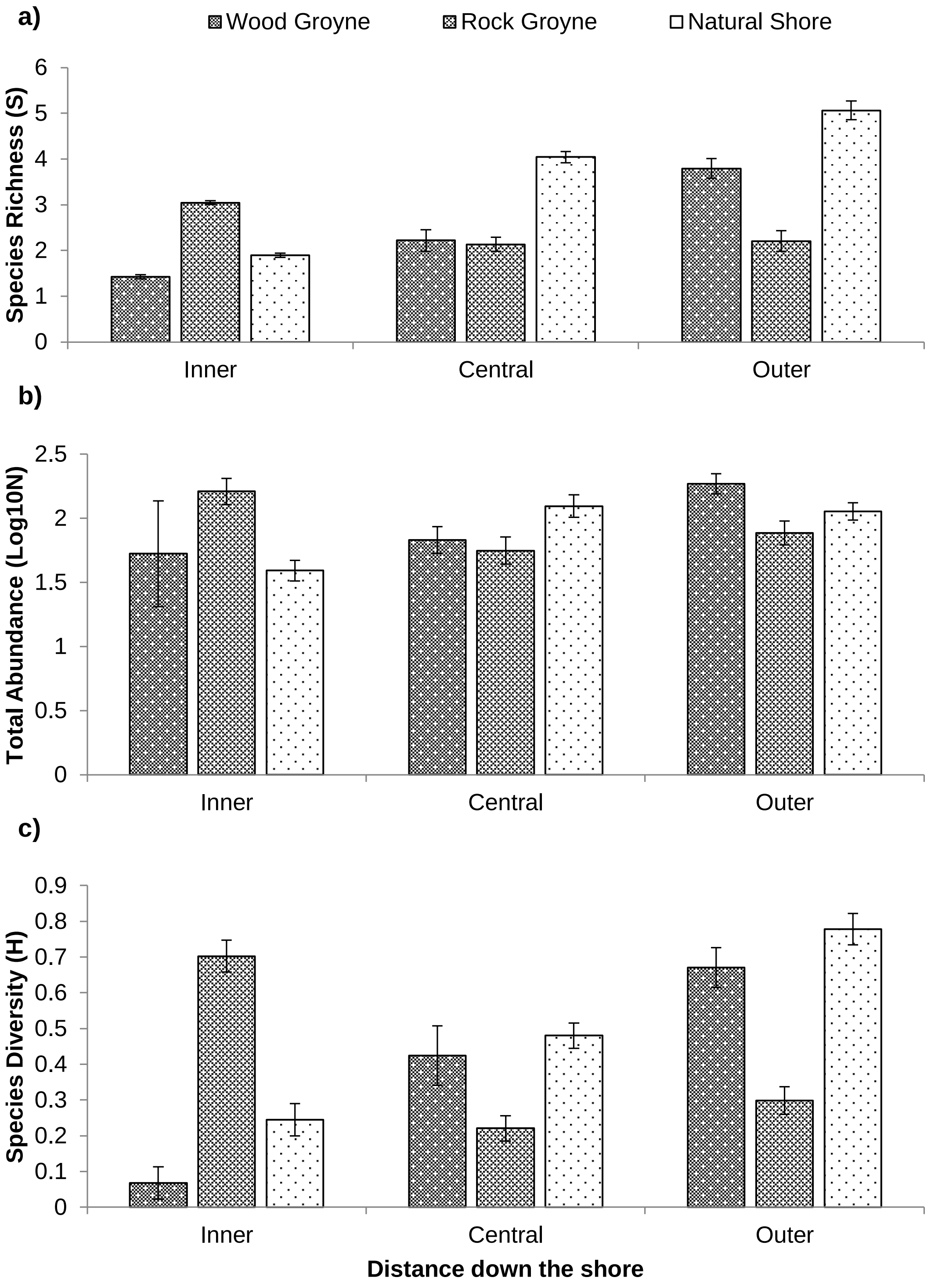

Figure 2.5 Variation in a) species richness (S), b) total abundance (LogN) and c) species diversity (H) with varied distance down the shore for the upper and mid zones on wooden groynes, rock groynes and natural shores (mean across all regions, +/- S.E).

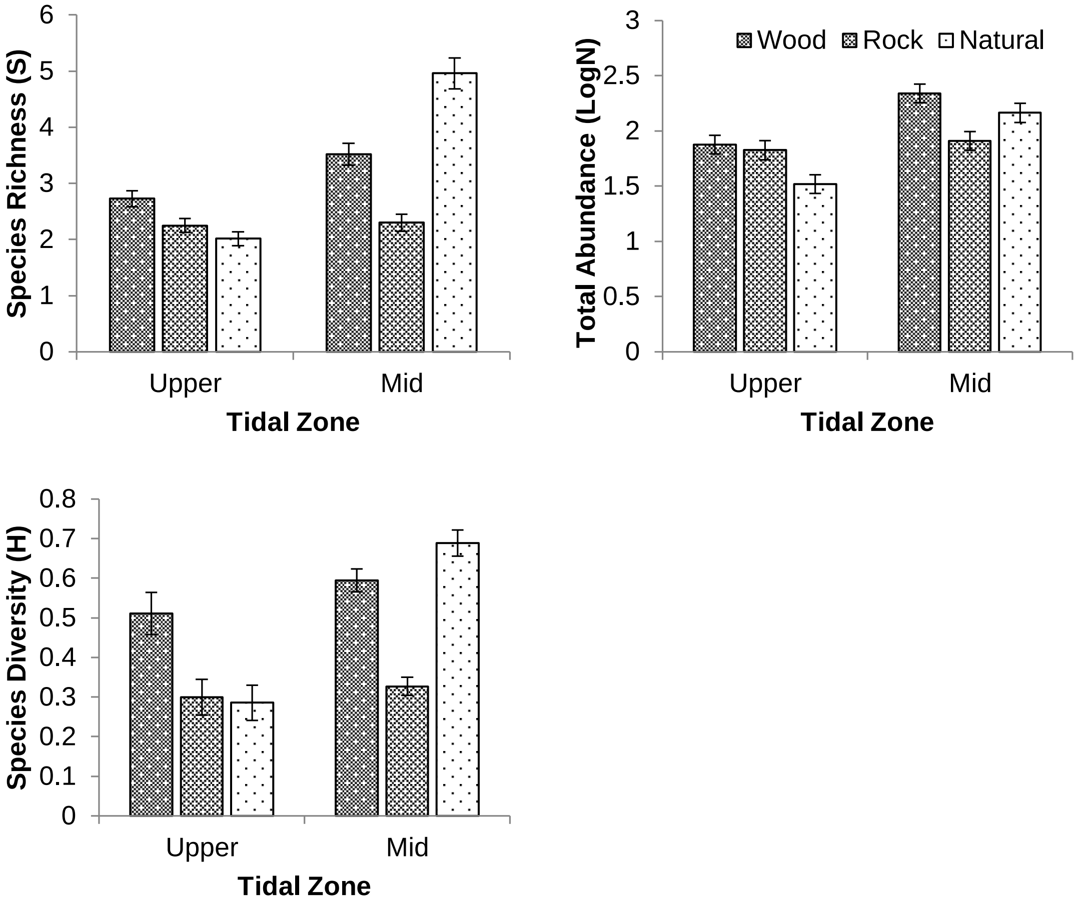

Figure 2.6 a) Mean species richness (S) , b) mean total abundance (logN), c) mean species diversity (H) of wooden groynes, rock groynes and natural shores for upper and mid tidal zones, averaged across all regions (+/- SE).

2.3.4 Wooden Groynes versus Rock Groynes: Substrate

The species richness, total abundance and species diversity were significantly greater on the wooden groynes in comparison to the rock groynes (Table 2.4). Across the four regions in total, 20 species were recorded on the wooden groynes and 18 species were recorded on the rock groynes. The SIMPER revealed the six species which accounted for 90% of the overall 88.6% dissimilarity between communities (Table 2.5) and the Canonical Analysis of Principal coordinate (CAP) illustrates the effects of substrate on specific species (Figure 2.7). The CAP has highlighted species which are more prevalent on either wooden or rock groynes based on a constrained ordination.

Table 2.4 Linear mixed models for a) species richness (square root scale), b) total abundance (Log scale), c) species diversity (H) in relation to environmental variables- Substrate, Aspect, Distance and Zone (Artificial Sites only, ***=P<0.001, **=P<0.01, *=P<0.05, NS= P>0.05).These LMM include terms for any significant interactions between any two factors which were statistically significant.

a) Species richness (S)
Model sqrt(S)~Substrate+Aspect+Distance+Zone+Distance*Zone+Aspect*Distance +1|Region/ TG)

Fixed Effects	**Estimate**	**Standard Error**	**t**	**P**
Intercept	1.523	0.178	8.562	**
Substrate Wooden	0.338	0.061	5.485	***
Aspect West	-0.078	0.044	-1.760	0.07
Distance Central	-0.273	0.061	-4.441	***
Distance Inner	-0.588	0.159	-3.696	***
Zone Upper	-0.222	0.047	-4.711	***
Distance Central: Zone Upper	0.163	0.075	2.144	*
Distance Inner: Zone Upper	0.345	0.180	1.912	0.06
Aspect West: Distance Central	-0.001	0.067	-0.026	0.97
Aspect West: Distance Inner	0.450	0.133	3.385	***
Random Effects	**Variance**	**Standard deviance**		
Transect (Site: Region)	0.014	0.119		
Region	0.115	0.339		

b) Total abundance (N)
Model LogeN~Substrate+Aspect+Distance+Zone +(1|Region/TG)

Source	**Estimate**	**Standard Error**	**t**	**P**
Intercept	4.104	0.893	4.595	*
Substrate Wooden	0.936	0.445	2.108	*
Aspect West	0.030	0.125	0.246	0.80
Distance Central	-0.522	0.133	-3.909	***
Distance Inner	-0.887	0.260	-3.407	***
Zone Upper	-0.558	0.132	-4.221	***
Random Effects	**Variance**	**Standard deviance**		
Transect (Site: Region)	1.023	1.011		
Region	2.741	1.655		

c) Species diversity (H)
Model H~Substrate+Aspect+Distance+Zone+Distance*Zone+Aspect*Distance+Substrate*Aspect+(1|Region/TG)

Fixed Effects	**Estimate**	**Standard Error**	**t**	**P**
Intercept	0.287	0.153	1.873	0.14
Substrate Wooden	0.380	0.067	5.659	***
Aspect West	0.036	0.057	0.629	0.53
Distance Central	-0.151	0.062	-2.451	*
Distance Inner	-0.282	0.160	-1.765	0.78
Zone Upper	-0.182	0.047	-3.848	***
Distance Central: Zone Upper	0.146	0.076	1.919	0.55
Distance Inner: Zone Upper	0.344	0.181	1.896	0.58
Aspect West: Distance Central	-0.063	0.067	-0.934	0.35
Aspect West: Distance Inner	0.469	0.135	3.440	***
Substrate Wooden: Aspect West	-0.133	0.066	-1.997	*
Random Effects	**Variance**	**Standard deviance**		
Transect (Site: Region)	0.014	0.120		
Region	0.082	0.286		

Table 2.5 SIMPER table indicating average abundance of species per 0.25m^2 quadrat on wooden groynes and rock groynes (Ave=Mean, Cum %= Cumulative percentage contribution to dissimilarity).

SIMPER: Rock & Wood groynes			**Ave dissimilarity= 88.60**		
Species	Rock Ave Abundance	Wood Ave Abundance	Ave Dissimilarity	Contribution %	Cum %
Semibalanus balanoides	202.47	409.56	37.88	42.76	42.76
Ulva spp.	11.49	31.95	14.59	16.47	59.23
Fucus spiralis	6.95	15.39	8.48	9.57	68.8
Austrominius modestus	1.86	159.86	7.38	8.33	77.13
Patella vulgata	14.13	2.34	6.56	7.41	84.54
Mytilus edulis	0.2	40.04	5.14	5.8	90.34

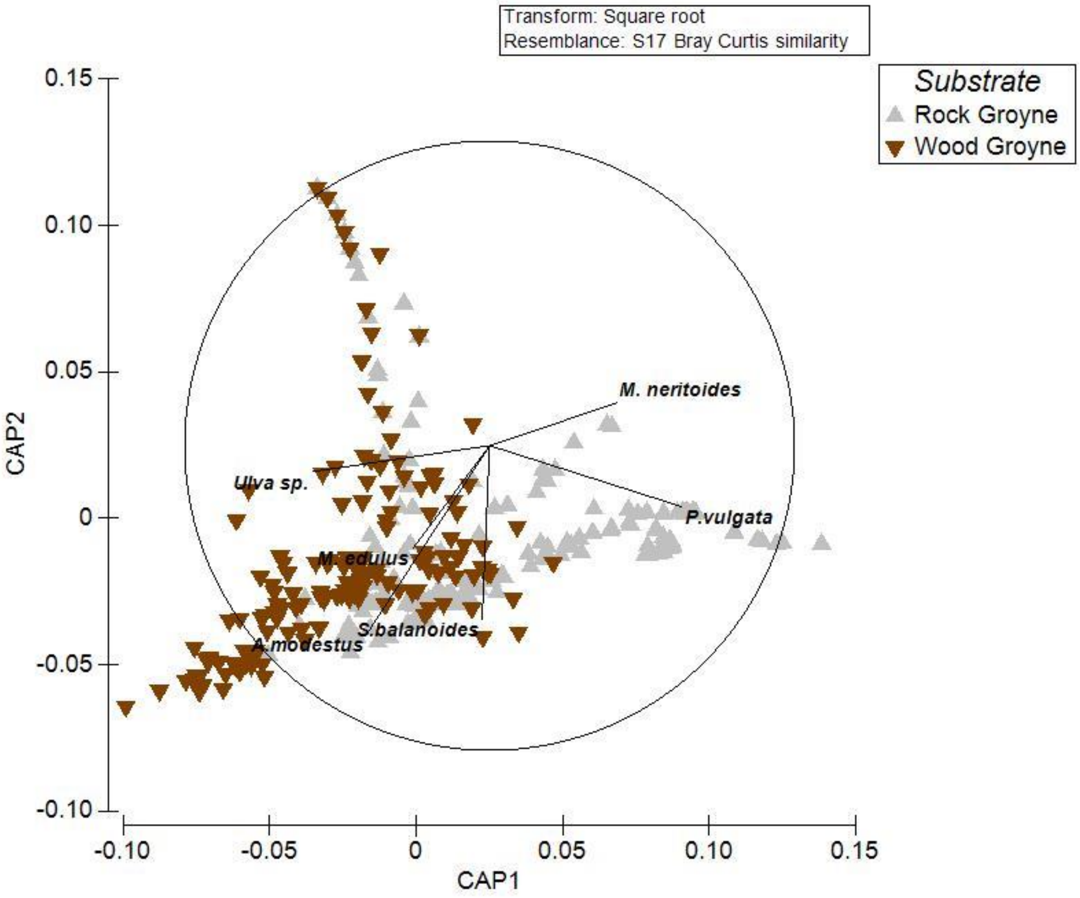

Figure 2.7 A Canonical Analysis of Principal coordinate (CAP) showing the effects of Substrate on communities of groynes only. Data from across all regions (Spearman's rank correlation of species vector overlaid (r^2 <0.4) (trace statistic= 2.46, P < 0.01), based 9,999 permutations.

2.3.5 Wooden groynes vs. Rock groynes: Aspect

When comparing the influence of aspect on the groynes, there was no significant difference in species richness, total abundance and species diversity of organisms on the west side of the wooden groynes compared to the east side. However, there was a significant interaction between Aspect and Distance down the shore for species richness and species diversity, specifically with the west inner sections of the groynes having the highest richness and diversity (Table 2.4). The temperature recorded on the east side of the groynes was significantly lower than on the west side of the groynes (east mean (SE) wood=19.1^{o}C (0.71), rock= 18.1 (0.65); west mean (SE) wood=25.46 (2.1), rock= 20.3 (1.3)) as illustrated in Figure 2.8. Temperature differences were greatest on the wooden groynes, independent of the time of day (morning/evening).

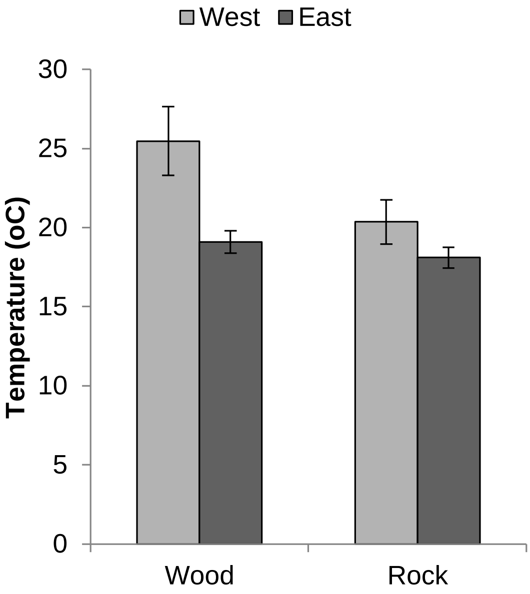

Figure 2.8 Mean temperature (^{o}C) on the east and west sides of wooden and rock groynes across all regions (+/- S.E.).

2.3.6 Wooden vs. Rock Groynes: Distance

Overall, there was a significant difference between the wooden and rock groyne species richness and diversity at comparable distances down the shore (Figure 2.5,

Table 2.5). The wooden groynes have significantly greater species richness and species diversity at the outer sections of the groynes, whereas the rock groynes showed the greatest species richness and diversity at the inner sections (Figure 2.5, Table 2.5).

2.3.7 Wooden vs. Rock Groynes: Zone

The tidal zone had a significant effect on the total abundance of organisms on wooden and rock groynes, with greater abundances on the mid shore sections (Figure 2.6, Table 2.5). The SIMPER analysis revealed that the upper sections of the rock groynes were dominated by *S. balanoides, Ulva* spp*., M. neritoides* and *P. vulgata*, whereas the upper sections of the wooden groynes were dominated by *Ulva* spp., *F. spiralis* and *S. balanoides.* The mid sections of the rock groynes were dominated by three species: *S. balanoides, P. vulgata* and *Ulva* spp., whereas the mid sections of the wooden groynes were dominated by five species *S. balanoides, Ulva* spp*., A. modestus, M. edulis* and *F. spiralis.*

2.4 Discussion

Overall, some artificial shores were more species poor and less diverse than the adjacent natural rocky shores, although there was no significant difference. These findings were consistent with Hypothesis 1 and other studies which have also found lower species richness and diversity on artificial structures (Bulleri & Chapman, 2004; Vaselli et al., 2008; Firth et al., 2016a). The wooden groynes supported a community most similar to the natural shores and, compared to rock groynes, had higher mean species richness and species diversity, contrary to Hypothesis 2. The wooden groynes also displayed similar patterns to the natural shores, with increased species richness and diversity with distance down the shore and lower tidal zones. Overall, the three most important factors in determining the community composition were Substrate Distance and Tidal Zone.

Without further experimentation, the individual effects of the materials used to construct the groynes cannot be separated from the structures themselves. However, substrate was a significant factor contributing to the differences in communities, supporting results from previous studies (Connell & Glasby, 1999; Herbert & Hawkins, 2006). Although no known previous research has compared wooden and rock groynes, studies on different rock types have found surface texture and mineralogy to be important factors in determining species recruitment and community composition (Caffey, 1982; Ambrose & Swarbrick, 1989; Holmes et al., 1997; Herbert & Hawkins, 2006). With the exception of groynes and piers, wood is typically absent from the marine environment in temperate regions, particularly in intertidal areas. In Sydney, Australia, Glasby et al., (2007) found that wooden pilings had lower recruitment rates of native species compared with concrete and sandstone. Yet, in the current study, there was no significant difference in the number of non-native species on the wooden and rock groynes, or between these structures and the natural rocky shores. Rock groynes are generally characterised by a lack of water retention as the substrate is relatively impervious, whereas wooden groynes can absorb some moisture. Moreover, the higher cover of fucoid algae, which was more prevalent on the wooden groynes, can create damp refugia at low tide (Figure 2.1a). On natural shores, rock pools and microhabitats provide a refuge from both physical and biological factors, such as wave exposure and predation (Underwood & Jernakoff, 1984; Benedetti-Cecchi & Cinelli, 1995), creating niches for a wider variety of organisms (Raffaelli & Hughes, 1978). Yet the substrate on the majority of artificial structures found in this study were relatively smooth and without rock pools. This lack of habitat heterogeneity is most likely responsible for the lower species richness observed overall.

The majority of artificial structures have steep vertical faces, whereas natural shores in the study areas had shallow sloping gradients. This variation in profile could have affected the composition and functioning of the communities through changes to foraging behaviour or frequency of dislodgement from wave action (Hawkins & Hartnoll, 1982; Hills & Thomason, 1996; Firth et al., 2014a). Compared to natural shores in the study area, groynes extend to a greater vertical height lower down the shore and can create relatively novel species distributions. The creation of 'height' in the outer zone allowed both high-shore species to survive further down the shore and typically low-shore species to survive at higher levels. Mussels (*M. edulis*) were generally found in the lower levels on these natural shores but, due to the creation of groynes in the mid-shore, they survived successfully at a higher level. Their persistence on the groynes could be due to the frequent spray created when the waves hit the groynes preventing desiccation at low tide and greater protection from predators such as crabs. As with wave-exposed natural shores, the groynes had a high total abundance of individuals but a low number of species. Small littorinid periwinkles (*M. neritoides)* were recorded on the outer section of the rock groynes, whereas on the natural shore they were only observed on the inner and central sections. In this regard, these habitats can resemble outer reefs on wave-exposed shores, where similar distribution patterns can be observed (Lewis, 1964). If species are in contact with water for longer periods they have the potential to release a greater amount of propagules. This could result in a higher settlement at habitats which were previously unconnected, resulting in changes in species distribution and facilitation of the spread of non-native species and climate migrants (Shanks et al., 2003; Kinlan et al., 2005).

The effect of Region on this study was greatest when comparing the wooden and rock groynes without the natural rocky shore comparisons, in which total abundance was specifically influenced by region as a random effect. Region 3, Worthing had lower species richness, total abundance and species diversity on the artificial structures than all other regions. Each region has its own coastal processes including tidal ranges, and physical processes such as sediment transport (Motyka & Brampton, 1993). The tidal patterns in Region 1 consist of a double high and low water with the smallest tidal range in the UK (2m) which restricts the intertidal habitat, therefore any additional habitat created will become quickly colonised. The tidal range increases from the Isle of Wight eastwards and from Portland Bill westwards increasing the intertidal habitat available. The regions may also differ in biology as variation in hydrodynamic conditions could alter the larval supply within each region and in turn the community composition.

Substrate temperature was found to be lower on the east facing side of the groynes compared to the west facing side of the groynes, with the greatest difference been recorded on the wooden groynes. The warmer conditions on the west side of the groynes could provide suitable settlement sites for species which are more heat tolerant such as the climate migrants *G. umbilicalis* and *P. depressa.* There has been a recorded eastward movement of southern intertidal invertebrate species in the UK (Mieszkowska et al., 2005) and the addition of artificial structures such as groyne may provide stepping stones for further expansion under the correct conditions (Moschella et al., 2005; Keith et al., 2011).

Connell and Glasby (1999) suggested that differences in assemblage composition on sandstone reefs and artificial structures were explained by the age of the community, which is fundamentally different between artificial structures and natural shorelines. As artificial structures were constructed at different times, they too have varied ages which will affect their community assemblages (Pinn et al., 2005). Many coastal structures are prone to regular maintenance and removal which may arrest community succession that accentuates differences with natural shores (See Figure 2.1b). Temporal variation in exposure of fresh surfaces also affects community composition, as the sequence of succession after disturbance is dependent on larval availability (Minchinton & Scheibling, 1991; Benedetti-Cecchi & Cinelli, 1996; Benedetti-Cecchi, 2000a; Hughes et al., 2000). Although samples were taken over two years, weather patterns were similar each year and there was no noticeable difference in recruitment or mortality of common species.

The natural shores had greater abundances of mobile grazers, such as *P. vulgata, L. littorea* and *G. umbilicalis,* whereas the groynes were dominated by sessile species such as *Ulva* spp. and *S. balanoides*. The low abundance of grazers on wooden groynes could explain the higher prevalence of large algae, specifically *Fucus spiralis* (Jenkins et al., 1999a, 1999b; Moore et al., 2007). Macro-algal canopies are known to influence community structure due to the sweeping effects of their fronds, resulting in dislodgement of mobile fauna and reduced settlement rates for barnacles (Hawkins, 1983; Jenkins et al., 1999b, 1999c). Grazers can be highly influential in controlling species distributions and abundance within the mid-tide region on natural shores (Raffaelli & Hawkins, 1996). The scarcity of benthic invertebrate predators on the groynes was also detected, with *Nucella lapillus* being the only species recorded. However, predation on benthic assemblages by fish and crabs could occur when groynes are submerged, and additionally by birds, such as herring gulls (*Larus argentatus*) and oystercatchers (*Haematopus ostralegus*) when the tide retreats.

The inside faces of the rock groyne boulders were not surveyed due to constrained access, resulting in a potential underrepresentation of cryptic species (Sherrard et al., 2016). Mobile predators, including larger fish and crabs, could not be quantified as the surveys were conducted at low tide. Seasonal variations in the communities could not be determined due to logistical constraints of weather and tide. Whilst both types of groynes did support non-native species (the barnacle *A. modestus*, Pacific oyster *Crassostrea gigas* and green alga *Codium fragilie)*, disproportionately high numbers of non-native species were not detected on the groynes, which is in contrast to a number of other studies on artificial structures (Vaselli et al., 2008; Mineur et al., 2012; Airoldi et al., 2015). Several climate migrants were recorded on both the natural shores and groynes, including *Perforatus perforatus*, *P. depressa* and *G. umbilicalis.* In areas where hard substrate is absent, groynes could thus act as stepping-stones for further range expansions (Moschella et al., 2005; Keith et al., 2011). The meta-population dynamics and connectivity between habitats of both natural and artificial is important to identify (Jonsson et al., 2016).

In summary, understanding species which commonly colonise artificial structures, such as groynes, is important for identifying anthropogenic influences on assemblages, and developing knowledge on community dynamics and the potential connectivity between habitats. This study found that although the artificial habitats were less diverse and species rich, they were still being colonised by marine species that formed assemblages and communities. Contrary to the hypotheses, the wooden groynes supported assemblages most similar to the natural shores. Whilst artificial structures appear to be homogenous, this study revealed they have high habitat heterogeneity at regional scales due to variation in substrate structure design, age, scouring and position on the shore. Not all coastal artificial structures provide diverse habitats and thus this variability can be incorporated into marine planning in order to optimise their ability to provide complex substrates that support diverse assemblages, as well as completing their functional roles.

3 The Recruitment and Mortality of Key Intertidal Species on Wooden and Rock Groynes

Abstract

Coastal defence structures are increasing worldwide and these structures have shown to support assemblages with lower diversity than natural shores. Substrate type and exposure is known to influence the establishment and diversity of intertidal communities on natural shores; however this has not been investigated on artificial structures. This study aims to investigate the effects of substrate on the settlement, recruitment and mortality of three habitat forming species on wooden and rock groynes. A comparative study of the recruitment of the barnacle *Semibalanus balanoides,* the limpet *Patella vulgata* and the mussel *Mytilus edulis,* on wooden and rock groynes in two regions on the south coast of England was compared to natural rocky shores over two years, 2015 and 2016. Overall there was distinct regional variation in *S. balanoide*s settlement and recruitment. Wooden groynes had the highest settlement overall, yet when regions were separated, site-specific variation was observed. Post recruitment mortality was greatest on the wooden groynes yet there was found to be no significant difference in post recruitment mortality on the sheltered or exposed side of the groynes. Populations of *P. vulgata* showed consistent levels of recruitment on both the natural shores and artificial structures, with the exception of the wooden groynes in Region 2 which had no recent recruitment. The age structure of *M. edulis* populations on wooden and rock groynes also showed stable recruitment over the two years with no gaps in age classes. This study helps determine the effect the groynes can have on ecological processes involved in establishing communities on artificial structures, in addition to highlighting the potential input these artificial structures might have into the metapopulations of the region.

3.1 Introduction

The proliferation of artificial coastal defence structures around the world is likely to influence natural settlement and recruitment processes of marine species. Habitat loss and increased fragmentation due to the presence of artificial structures is affecting population connectivity and altering coastal ecosystems (Airoldi & Beck, 2007; Goodsell et al., 2007; Bulleri & Chapman, 2010; Perkins et al., 2015). Changes to local and regional hydrodynamic processes can be associated with artificial structures which in turn could affect the larval supply of populations to both natural and artificial shorelines (Zanuttigh et al., 2011; Becchi et al., 2014; Dong et al., 2016; Scott et al., 2016). Artificial structures have acted as stepping stones for dispersal of native and non-native rocky biota in soft sediment habitats (Bulleri & Airoldi, 2005; Vaselli et al., 2008) and the substrates used to construct these features can greatly affect the chances of colonisation and subsequent community development (Bracewell et al., 2013). Artificial structures can form islands of hard substrate within soft sediment environments which are susceptible to the laws of island biogeography theory (Chapter 1 Section 1.3). The size and quality of the habitat along with the connectivity between habitats are major drivers in the colonisation and stability of populations (MacArthur & Wilson, 1967). By understanding the ecological processes of species settlement, recruitment and mortality involved in the development of these communities we can predict the composition of assemblages on different substrata and structures.

Settlement

Marine benthic organisms can reproduce both sexually and asexually and the release of pelagic larvae and spores is common with rocky shore species (Little, et al 2009). Larvae have variable pelagic durations and some species can travel hundreds of kilometres before attachment and metamorphosis (Shanks et al., 2003; Jenkins et al., 2009; Shanks, 2009). Settlement is defined as the process in which contact is made with the substratum which includes exploratory behaviour, orientation and attachment

(Connell, 1985; Pawlik, 1992). Various factors influence the chances of larval settlement, including type of substratum, hydrodynamics and larval availability (Jenkins et al., 2000; Underwood & Keough, 2001). The density at which settlement occurs is also known to be dependent on small scale physical surface characteristics such as colour and illumination of the substratum (Kon-ya & Miki, 1994) and water flow rates (Knights et al., 2012) alongside biological factors such as algal cover (Hawkins, 1983), biofilm development (Henschel & Cook, 1990), presence of conspecifics (Knight-Jones, 1953) and available free space (Minchinton & Scheibling, 1993). In addition the age of the larvae is also known to influence the likelihood of settlement; the older the larvae the greater the chance of settling on a poor habitat due to the "desperation hypothesis" (Botello & Krug, 2006; Elkin & Marshall, 2007).

Recruitment and Mortality

Recruitment is the number of metamorphosed larvae at the end of the settlement period (Jenkins et al., 2000). Recruitment is a combination of settlement and post settlement mortality which occurs up until the first census. The post-settlement period of marine benthic invertebrates has the highest mortality rate, with 90% being recorded in some cases for bivalves, gastropods, barnacles and ascidians (Gosselin & Qian, 1997). Harsh post settlement abiotic factors include desiccation (Branch, 1975; Denley & Underwood, 1979), wave action (Vadas et al., 1990) and emersion (Raffaelli & Hawkins, 1996) alongside biotic factors such as grazing (Hawkins & Hartnoll, 1985), algal sweeping (Jenkins et al., 1999b, 1999c), and "bulldozing" by grazers (Miller, 1986; Raffaelli & Hawkins, 1996). Recruitment and mortality can also be affected by substrate type (Raimondi, 1988; Holmes et al., 1997; Herbert & Hawkins, 2006). On artificial structures some of these conditions are modified, for example, groynes have two aspects with varying wave exposure, which could affect the survival of larvae/ juvenile individuals (Underwood & Jernakoff, 1984; Pinn et al., 2005). The overall population size is dependent on both the number of larvae arriving at a site and the percentage of surviving recruits which achieve adulthood (Little, et al., 2009).

Key species of rocky shores in the North Atlantic

Semibalanus balanoides has a widespread distribution in the UK and typically inhabits shores of varying exposure within the mid tide zone. Barnacles are key space occupiers which form distinct zones on most intertidal shores (Ballantine, 1961a). *S. balanoides* is a cross-fertilising hermaphrodite and copulation occurs between November and December after which the fertilised embryos are incubated over winter (Rainbow, 1984). Naupilli larvae which have a pelagic larval duration of ~ 28 days (Pineda et al., 2002) are released into the water column during spring (Feb-April) and larvae select a favourable settlement site through the process of broad exploration, close exploration and inspection (Jenkins, et al 2009). At the inspection stage the larva rotates itself to a scale of its own body size to determine if there is adequate room for growth (Jenkins, et al 2009). Barnacles are gregarious and larvae are attracted to chemical cues produced by existing adult barnacles (Knight-Jones 1953). *S. balanoides* have been shown to settle on rough surfaces which have more potential microhabitats to form suitable settlement sites (Anderson & Underwood, 1994; Holmes et al., 1997; Herbert & Hawkins, 2006). Pomerat & Weiss (1946) found that porous and fibrous surfaces such as wood are colonised better than smooth non-porous surfaces. Increased porosity can reduce the levels of desiccation stress, improving the chances of survival and recruitment. Substrate temperature also influences desiccation stress, and darker coloured surfaces, which absorb more heat, may increase or limit the survival of recruits (Raimondi, 1988). The main predators of *S. balanoides* are dog whelks (*Nucella lapillus*), crabs (*Carcinus maenas*) and shannys (*Lipophrys pholis*) (Burrows et al., 1999; Mauck & Harkless, 2001).

Patella vulgata is the most common limpet on British shores and a “keystone” species as it is an important intertidal grazer in the high to mid shore. The removal of limpets enables the ephemeral green algae and fucoids to dominate shores and can consequently change the community structure of the habitat (Hawkins & Hartnoll, 1985; Hawkins & Southward, 1992; Raffaelli & Hawkins, 1996). Following the Torrey Canyon oil spill on the south west coast of England in 1967 the loss of *P. vulgata* resulted in fluctuations of dense fucoid canopies for approximately 11 years, after which the populations managed to recover to a sustainable level (Hawkins & Southward, 1992; Raffaelli & Hawkins, 1996).

P. vulgata releases eggs annually (October-January) which have a pelagic life of around two weeks and are fertilised in the water column (Ballantine, 1961b). Newly settled spat (2 mm) may be found in damp cracks and crevices until they reach 4-5 mm, at which point they move out onto drier rocks (Crump et al., 2003). Juvenile limpets are susceptible to high rates of mortality due to desiccation stress and they seek refuge in damp sheltered areas until they are large enough to withstand the conditions. Limpets are said to have reached adulthood after one year and individuals larger than 6 mm are thought to be adults (Little et al., 2009). Limpets have good defence mechanisms including a hard shell which it uses to bulldoze other limpets, yet they still have two main predators which are crabs and birds (Silva et al., 2008).

Mussels (*Mytilus edulis*) are bivalve filter feeders which can create extensive biogenic beds in both intertidal and subtidal areas. They have been shown to inhabit low shore artificial structures such as groynes (Bacchiocchi & Airoldi, 2003). Mussel beds provide secondary habitats for a high number of invertebrates such as annelids, crustaceans and echinoderms (O'Connor & Crowe, 2008; Seed, 2009). On European rocky shores, mussel beds are biogenic bed features that have a high conservation value (Holt et al., 1998). *M. edulis* reproduce twice a year during spring and summer through external fertilisation. As with barnacle larvae, settlement sites are tested before attachment; mussels have the ability to subsequently detach and secondary settlement can occur if the previous site was inadequate (Raffaelli & Hawkins, 1996). Primary mussel settlement is associated with filamentous surfaces, such as algae, including *Polysiphonia* spp. and *Ceramium* spp. or adult byssus threads (Seed, 1969; Bayne, 1976). Once they have reached 1-3 mm in size the mussel spat will then move onto suitable adult substrata. The main predators of *M. edulis* are dog whelks, starfish, crabs, flat fish and birds (Seed, 1969; Smith & Jennings, 2000). Fluxes in recruitment of mussels have previously been recorded (Wootton & Forester, 2013) and research has shown they can take a long time to recover after disturbance events (Suchanek & Suchanek, 1981; Brosnan & Crumrine, 1994).

Aims and hypotheses

Due to the lack of research into the population dynamics within communities associated with intertidal groyne structures and substrates, this study compares the settlement, recruitment and mortality of intertidal species on wooden and rock groynes. Focusing on three particular keystone and habitat-forming species; *S. balanoides, P. vulgata* and *M. edulis*, the importance of these processes are compared between artificial structures and natural rocky shores across two regions. This chapter will assess the stability of populations on artificial structures which may determine their regional importance within metapopulations and potential interaction with natural habitats. The aims of this chapter are (i) to determine if artificial substratum influences the settlement, recruitment and mortality of three key intertidal species; *S. balanoides, P. vulgata* and *M. edulis* and (ii) to determine if groyne type and aspect influence the mortality of *S. balanoides*.

The hypotheses to be tested are:

1) Natural shores will have higher levels of recruitment than groynes due to greater larval availability from adult populations.
2) Rock groynes will have higher rates of settlement of *S. balanoides* than wooden groynes due to the substrate similarities between rock groyne boulders and natural rock shore boulders.
3) Wooden groynes will have lower rates of mortality of S*. balanoides* than rock groynes as a result of being able to obtain a more secure attachment on the surface of the wood in comparison to the rock.

4) The most exposed sides of the groynes (west) will have highest mortality of *S. balanoides* due to increased wave exposure.
5) The population size frequency of *P. vulgata* and *M. edulis* will be more stable over time on natural shores in comparison to groynes due to the more established population on the natural shores.

3.2 Methods

3.2.1 Study Sites

The settlement, recruitment and mortality of *S. balanoides* and the recruitment of *P. vulgata* were investigated during 2015 and 2016. Studies were conducted in two regions on the south coast of England: Region 1-Poole/Purbeck and Region 2-Hampshire/Isle of Wight (IOW) (Figure 3.1). Each region had one artificial site (Bournemouth/Lee-on-Solent) which contained wooden groynes and rock groynes in close proximity and two natural rocky shore sites (Kimmeridge & Osmington/ Bembridge & Shanklin). Details on groyne substrate types and site descriptions are given in Chapter 2, Table 2.1. The recruitment of *M. edulis* was conducted on the wooden and rock groynes at the same artificial sites as above (Bournemouth & Lee-on-Solent), alongside the closest natural shore with *M. edulis* populations (Calshot). Calshot is located between the two artificial sites and the shore is comprised of mixed sediment where the mussel populations form aggregations in the mud/pebbles.

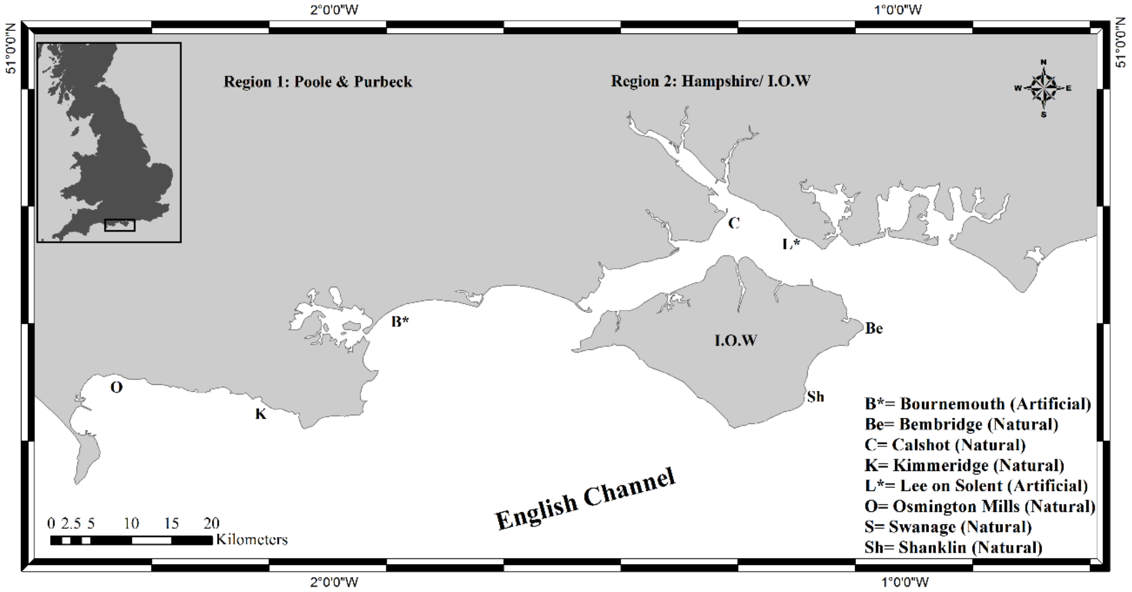

Figure 3.1 Location map for all sites within the two Regions. Bournemouth (Region 1) and Lee-on-Solent (Region 2) for wooden and rock groynes. Osmington Mills, Kimmeridge (Region 1), Bembridge, Shanklin and Calshot (Region 2) as natural sites (* indicates artificial shore which include both wooden and rock groynes).

3.2.2 Sampling design

Barnacle recruitment

Barnacle recruitment and survival was monitored within quadrat clearances on five wooden groynes, five rock groynes and two natural shores associated with each region. In February-March, prior to settlement of *S. balanoides*, in both regions, ten 100 cm^2 quadrats were scraped clear of all organisms on vertical surfaces on both sides (east/west) of 5 wooden and 5 rock groynes (Total N=100, 50 on each aspect). The cleared quadrats were located in the centre of the *S. balanoides* zone Mean Low Water (MLW) and marked with a screw in the top left hand corner to allow quick and accurate relocation. Quadrats were cleared using a paint scraper, wire brushes and blow torched to remove all microalgal films. In 2015, sites were revisited fortnightly for the first ten weeks and then at the end of September and March to record post- recruitment mortality during summer and winter respectively. In 2016 the quadrat clearances were repeated and monitored at the end of the settlement period (June), at the end of summer (September) and at the end of winter (March). Photographs were taken of each quadrat using a Nikon Coolpix AW120 and barnacles were counted using image analysis software ImageJ (Abràmoff et al., 2004).

Clearances on natural rocky shores were created at Osmington Mills, Kimmeridge, Bembridge and Shanklin. Fifty 100 cm^2 quadrats at each site were cleared on vertical surfaces prior to settlement and marked with a screw in the top left-hand corner. In 2015 photographs were taken of each quadrat fortnightly for the first 10 weeks then at the end of September and March to monitor settlement and post recruitment mortality. Again, clearances were repeated in 2016 and surveyed at the end of settlement period (June), at the end of summer (September) and at the end of winter (March). All sites were sampled within seven days of each other to enable an accurate comparison of settlement rates.

Limpet recruitment

In each region, the five wooden and the five rock groynes were surveyed during summer (June-August) over a two-year period (2015-16). The monitoring involved counting and measuring individual limpets within five haphazardly placed 0.25 m^2 quadrats on the vertical sides of the groynes at Mean Low Water. Measurements of shell length (to nearest mm) were taken of each individual limpet within the quadrat (Jenkins et al., 1999a). Care was taken on the rock groynes to choose independent boulders to avoid bias. At the control sites, Osmington Mills, Kimmeridge, Bembridge and Shanklin, a comparable methodology was used. Five 0.25 m^2 quadrats were placed on boulders or vertical sloping surfaces at MLW to count and measure the total shell length for each limpet with callipers to nearest (mm).

Mussel recruitment

Samples of mussels were taken from three wooden and three rock groynes at Bournemouth and Lee-on-Solent during June 2015 and June 2016. A 10cm diameter corer was pushed onto the vertical face of the groyne and the mussels were removed with a metal scraper. Mussel shells were measured using callipers (to nearest mm) for total length in order to determine age classes. In each region during 2015, 6 samples were collected at Mean Low Water on each type of groyne (3 of each aspect (east/west) of the wooden and rock groynes), and on natural shores. In 2016 to investigate more closely the variation in mussel size and density, 18 samples were collected per groyne and shore habitat (9 per each side/aspect on the groynes). During both years 6 samples were taken from the natural shore (Calshot), in which variation in aspect could not be tested.

3.2.3 Statistical analysis

To test the hypotheses, datasets from 2015 and 2016 were analysed separately. All data was tested for normality and equal variances and all tests were run using R v3.2.4 (R Core Team, 2016).

Barnacle recruitment and mortality

Statistical differences in *S. balanoides* recruitment were tested with factors Region, and Substrate, using a Quasi-Poisson Generalised Linear Model (GLM) (Bolker et al., 2009). Due to observations in count data this model resolved issues relating to over-dispersion and after examination of the residuals, this model was determined to be the most applicable to the data (Zuur et al., 2009).

The summer and winter post recruitment mortality for *S. balanoides* were calculated using the following equations:

Summer mortality= Ln (no. June +1) – Ln (no. September +1)

Winter mortality= Ln (no. September +1) – Ln (no. March + 1)

Statistical differences in *S. balanoides* post recruitment mortality were tested with factors Region and Substrate alongside an interaction between Substrate and Season (summer/winter), using a Quasi-Poisson GLM (Bolker et al., 2009). A separate GLM test was conducted to determine any statistical difference in *S. balanoides* mortality across the different aspects (east/west) on the wooden and rock groynes for each region (no natural control data).

Limpet & Mussel recruitment and mortality

Size frequency histograms were created for each year to determine age classes of limpets and mussels on wooden and rock groynes in comparison to natural shores.

3.3 Results

3.3.1 Barnacle settlement

The majority of *S. balanoides* settlement occurred between March and May, corresponding to previous studies (Southward & Crisp, 1956; Hawkins, 1982; Jenkins et al., 2001). The settlement curves for the 2015-2016 period show variation between regions, with Region 1 having a significantly higher rate of settlement on the wooden groynes compared with both natural shores and rock groynes (Figure 3.2). In contrast, in Region 2 there was a higher settlement on the rock and natural shores in comparison to wooden groynes (Figure 3.2).

Barnacle recruitment

In 2015 and 2016 the recruitment was significantly higher in Region 2 than Region 1 (Figure 3.3). In 2015 and 2016 the difference between Regions accounted for 38.5-43.1% of the overall variation found in the GLM ANOVA (Table 3.1) with the remaining variation accounted for by Substrate (5.14-6.15%) (Table 3.1). The pairwise tests revealed that in 2015 wood and rock groynes had significantly higher recruitment of *S.*

balanoides than the natural shores (Table 3.1, Figure 3.3), yet there was no difference between rock and wood. However in 2016 there was a significant difference in recruitment between all substrates, with rock groynes having the greatest recruitment, followed by wooden groynes; natural shores had the lowest recruitment overall (Figure 3.4). Therefore hypothesis 1 can be rejected and hypothesis 2 can be accepted.

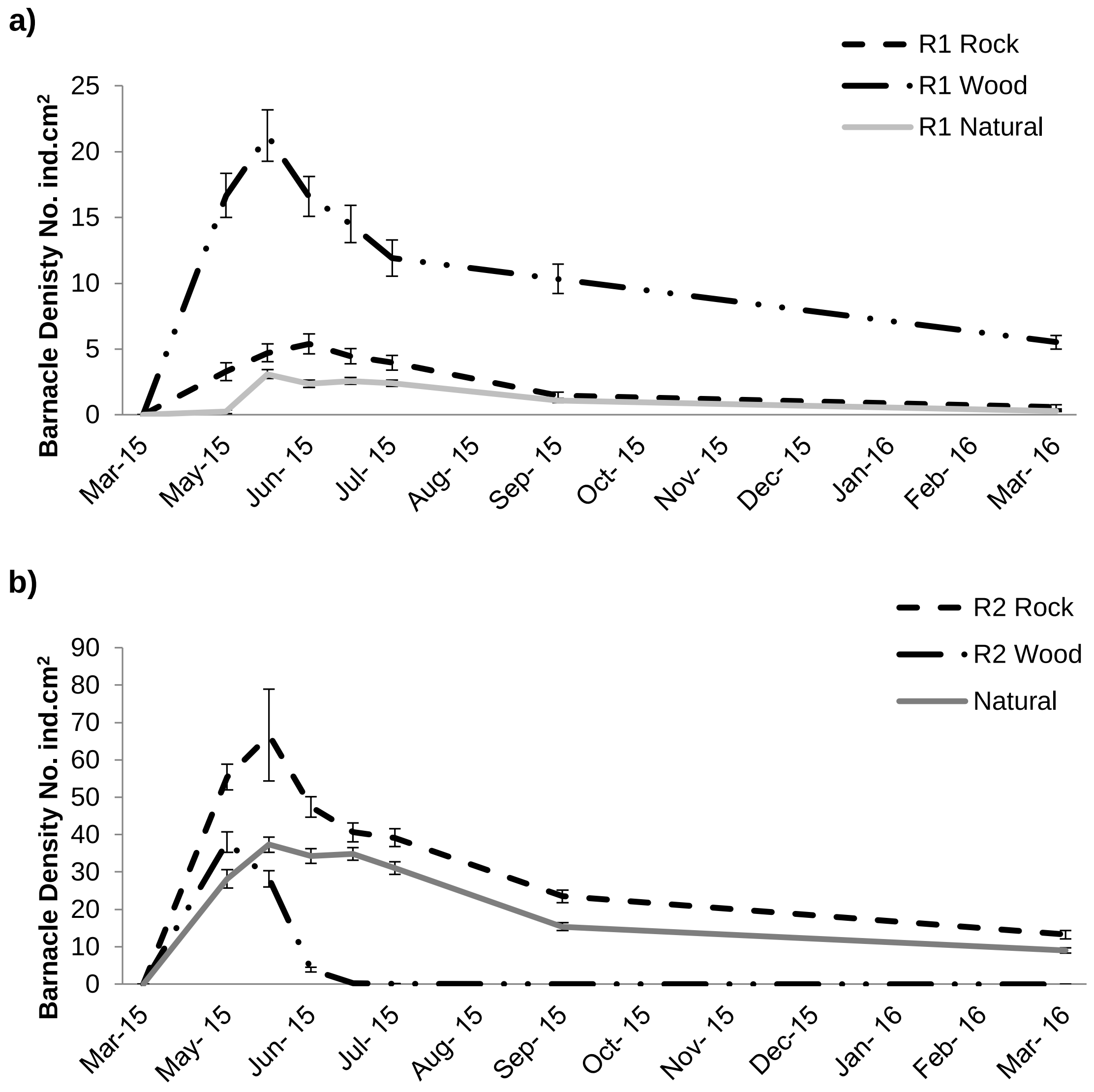

Figure 3.2 Mean *S. balanoides* density per cm^2 for the settlement period of 2015-2016 on rock, wood and natural substrate at a) Region 1- Poole/Purbeck and b) Region 2- Hampshire/IOW (Natural shores averaged, +/- S.E, note variation in axis.)

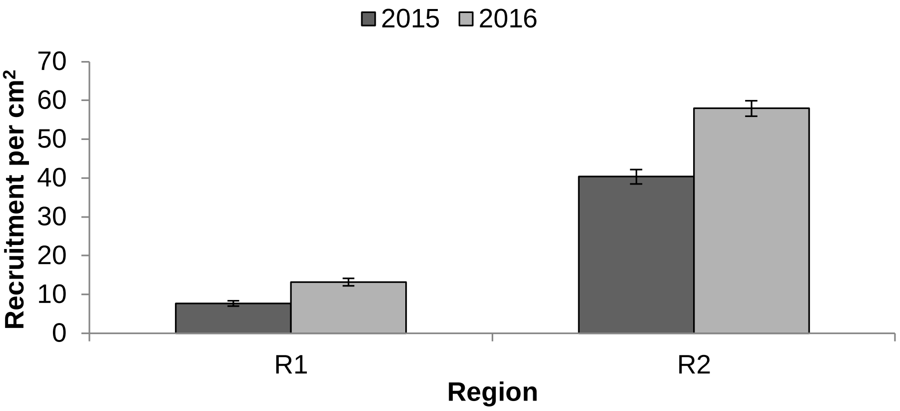

Figure 3.3 Mean recruitment of *S. balanoides* after the period of 1 year in June 2015 & 2016 across regions, R1=Poole/Purbeck, R2= Hampshire/IOW (n = 600, +/- S.E.).

Table 3.1 Analysis of variance Generalised Linear Model (GLM) of recruitment of *S. balanoides* in 2015 and 2016 between Regions and substrate (% explained is the % each factor contributed to the overall variance, ***=P<0.001, **=P<0.01,*=0.05 NS= Not significant).

2015	**df**	**Deviance**	**F value**	**P value**	**% Explained**
Region	1	72329	685.66	***	38.5%
Substrate	2	9666	45.81	***	5.14%
Region* Substrate	2	14038	40.61	***	7.47%
Contrasts: Substrate			**Z ratio**	**P value**	
Natural – Rock			-4.832	***	
Natural – Wood			-4.301	***	
Rock - Wood			0.588	NS	
2016	**df**	**Deviance**	**F value**	**P value**	**% Explained**
Region	1	83699	977.39	***	43.14%
Substrate	2	11948	69.73	***	6.15%
Region * Substrate	2	19385	62.72	***	9.99%
Contrasts: Substrate			**Z ratio**	**P value**	
Natural – Rock			-3.055	**	
Natural – Wood			4.727	***	
Rock - Wood			7.437	***	

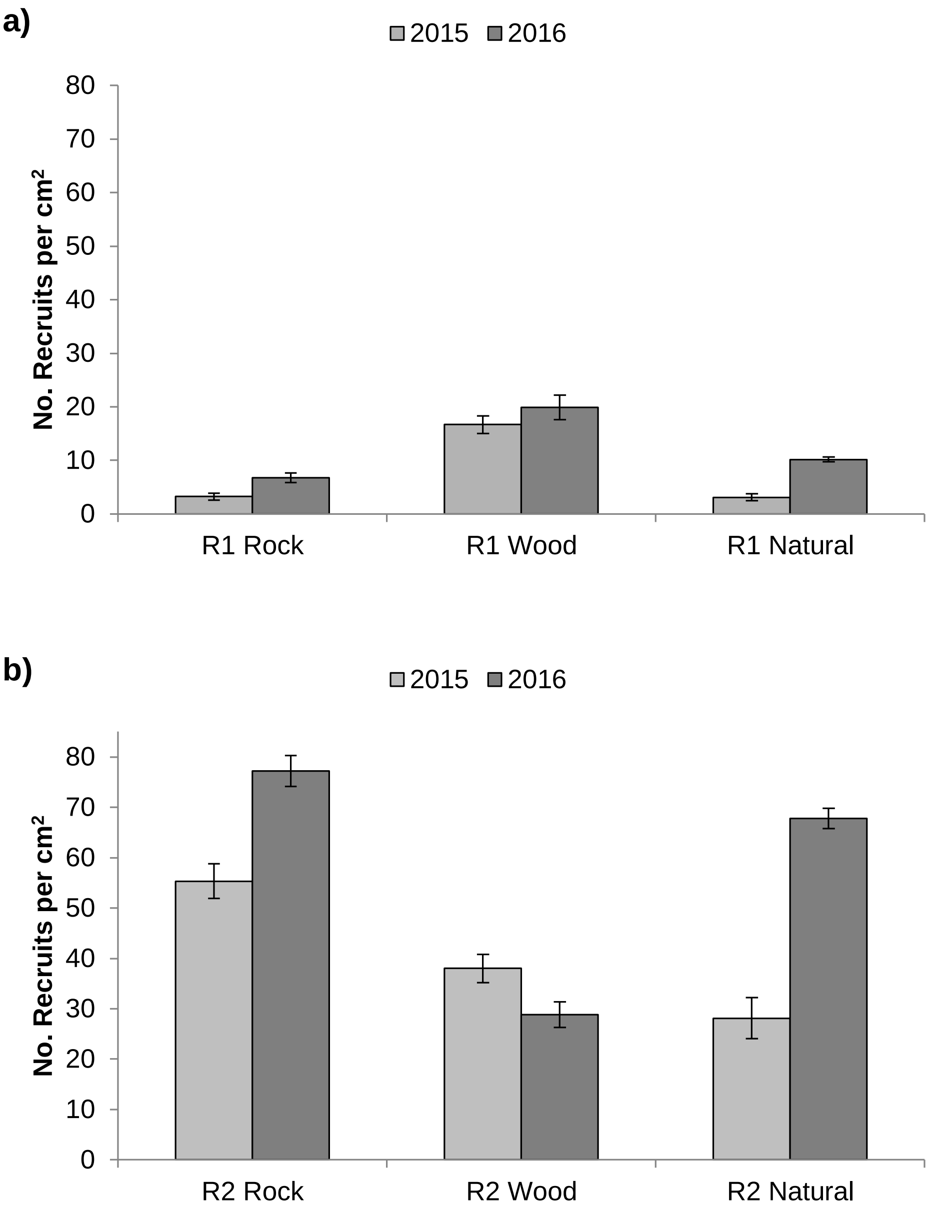

Figure 3.4 Mean recruitment of *S. balanoides* after the period of 1 year in June 2015 & 2016, variation between substrates- rock, wood and natural in a) Region 1 and b) Region 2 (Natural shores averaged, n=100, +/- S.E).

Barnacle Post Recruitment density

The post recruitment densities of *S. balanoides* during 2015 and 2016 showed variations between sampling regions indicating context dependant variation (Figure 3.2 & 3.3). Region 1 had higher densities on the wood groynes across both years, whereas Region 2 had higher densities on rock groynes over the two years (Figure 3.5).

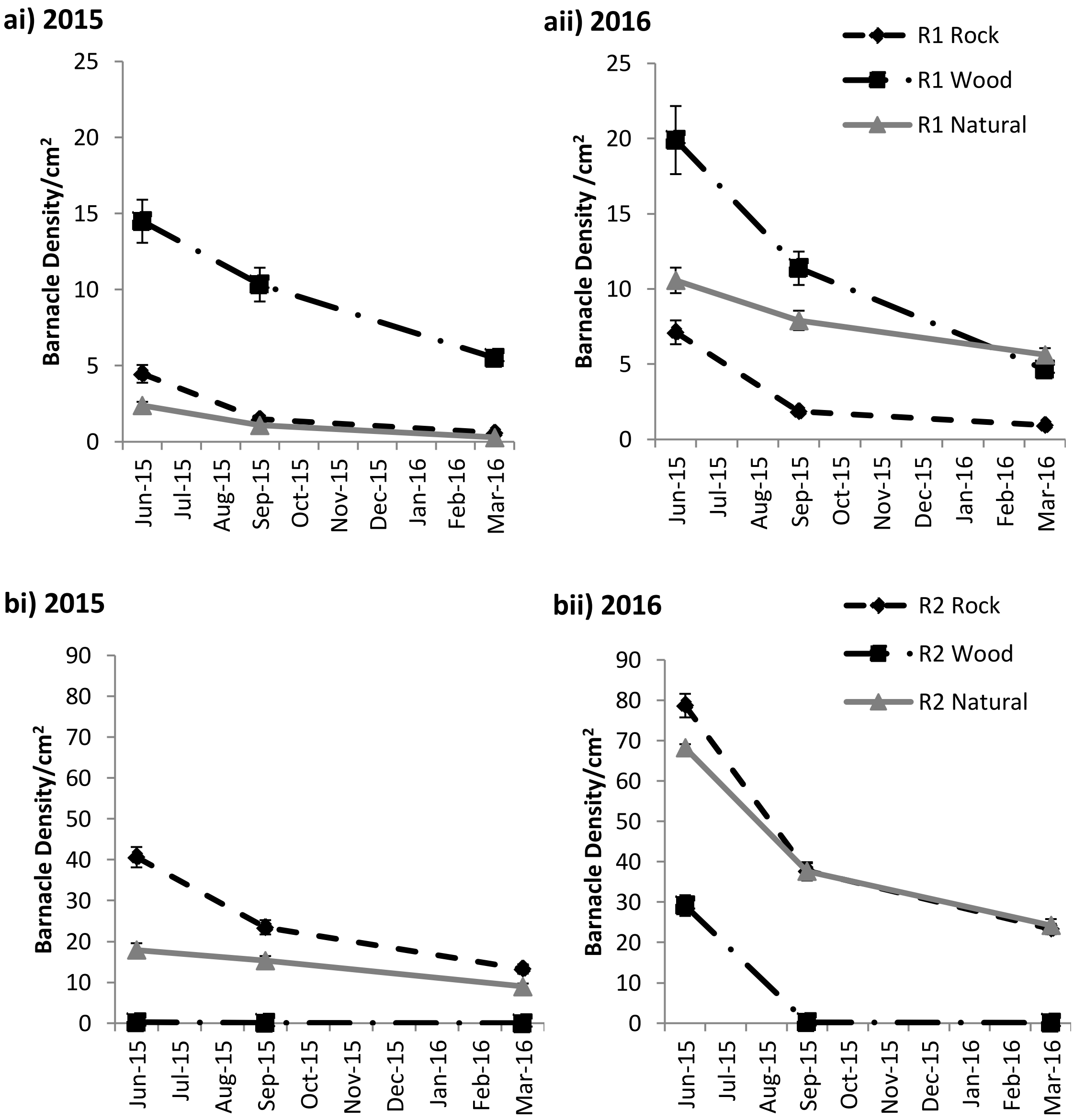

Figure 3.5 Barnacle density per cm^2 for rock, wood and natural substrates in a) Region 1- Poole/Purbeck and b) Region 2- Hampshire/IOW for i) 2015 and ii) 2016 (Natural shore is an average of both natural shores, n= 100, +/- S.E.). Note difference in y-axis scale between Regions.

Seasonal barnacle mortality

In 2015 and 2016 the mortality of *S. balanoides* was significantly dependant on substrate, season and region with the interaction between season and substrate accounting for the greatest proportion of the variation particularly in 2015 (Table 3.2). The pairwise comparisons revealed that the mortality on wood was significantly different to both rock and natural shores in both 2015 and 2016, however in 2015 no differences were found between rock and natural shores (Table 3.2). Overall there was a significantly higher mortality on the wooden groynes (Figure 3.6), yet the variation in region was due to the significantly greater summer mortality on the wooden groynes within Region 2 (Figure 3.7). Hypothesis 3 is rejected, however, the regionally specific variation has been acknowledged. In 2015 the winter mortality within Region 1 and 2 was greater than the summer mortality, with the exception of Region 2 wood groynes (Figure 3.7). Yet in 2016 there was a higher proportion of summer mortality across both regions with the exception of Bournemouth wooden groynes (Figure 3.7).

Table 3.2 Analysis of variance Generalised Linear Model (GLM) of mortality of *S. balanoides* in 2015 and 2016 between substrates at different regions and substrate, with an interaction between season and substrate. (% explained is the % each factor contributed to the overall variance ***=P<0.001, **=P<0.01,*=0.05 NS= Not significant).

2015	**df**	**Deviance**	**F value**	**P value**	**% Explained**
Substrate	2	229.17	88.85	***	9.97%
Season	1	225.78	175.08	***	9.82%
Region	1	114.73	88.97	***	4.99%
Substrate: Season	2	270.54	104.89	***	11.77%
Contrasts: Substrate			**Z ratio**	**P value**	
Natural – Rock			-1.88	NS	
Natural – Wood			-9.83	***	
Rock - Wood			-7.94	***	
2016	**df**	**Deviance**	**F value**	**P value**	**% Explained**
Substrate	2	185.17	51.43	***	8.90%
Season	1	172.53	95.84	***	8.29%
Region	1	84.22	46.78	***	4.05%
Substrate: Season	2	71.92	19.97	***	3.45%
Contrasts: Substrate			**Z ratio**	**P value**	
Natural – Rock			-3.54	**	
Natural – Wood			-9.16	***	
Rock - Wood			-5.08	***	

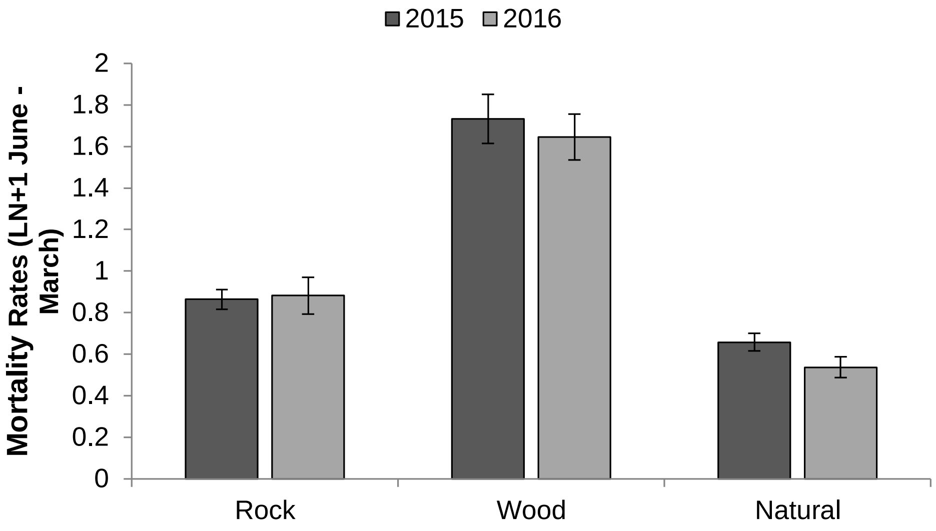

Figure 3.6 Mean post-recruitment mortality of *S. balanoides* (June-March) in 2015 and 2016 on rock groynes, wood groynes and natural shores, both regions combined (+/- S.E.)

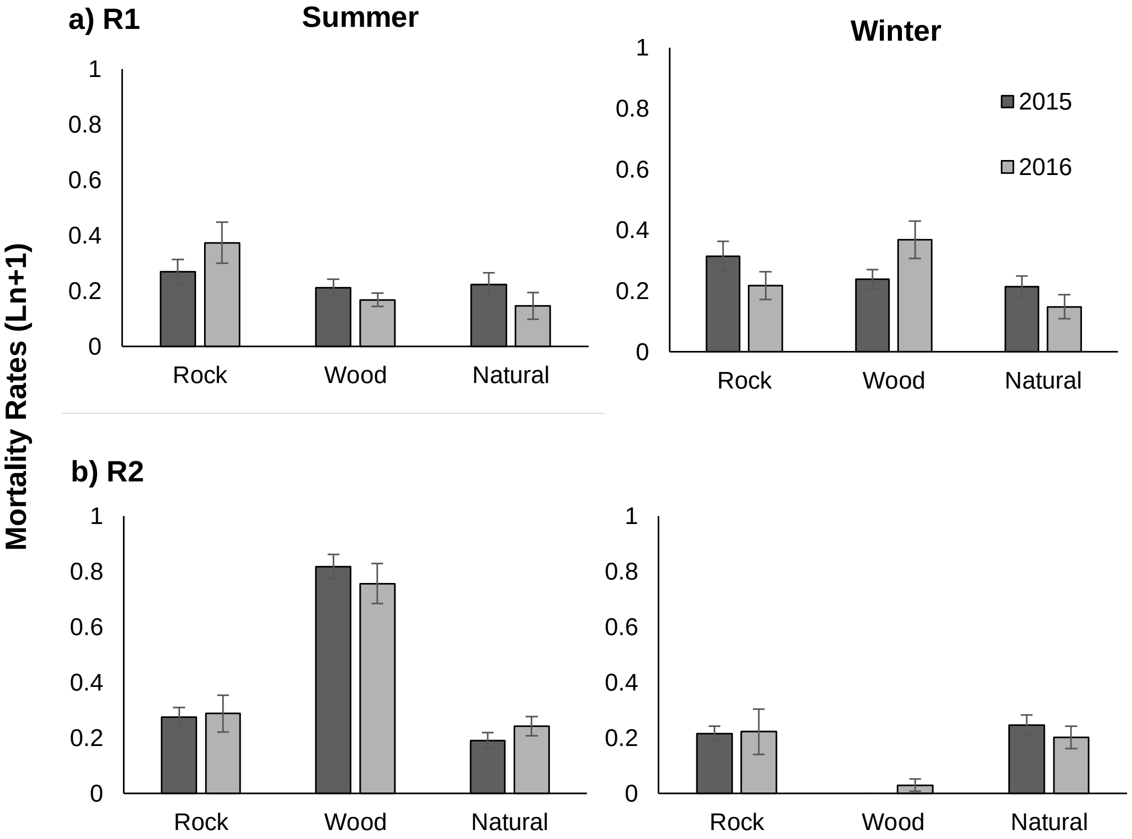

Figure 3.7 Mean post recruitment mortality of *S. balanoides* in summer and winter of a) R1- Poole/Purbeck and b) Region 2-Hampshire/IOW for 2015 and 2016. Summer mortality= Ln (no. June +1) – Ln (no. September +1) & Winter mortality= Ln (no. September +1) – Ln (no. March + 1). (Mean of 2 natural sites, +/- S.E.)

Aspect

In 2015 and 2016 there was no significant difference in mortality in relation to aspect on either the wooden or rock groynes within either region (P>0.05 in both years), therefore Hypothesis 4 can be rejected.

3.3.2 Limpet recruitment and survival

In Region 1 there was variation in *P. vulgata* size frequency between substrates and across years (Figure 3.8). In 2015 the rock groynes had a wider range of sizes 6-54 mm) when compared with the wooden groynes (6-46 mm). Yet in 2016 the larger sized limpets were absent on the rock groynes leaving a smaller size range (2-36 mm) whereas the limpets on the wooden groynes remained fairly similar (6-50 mm) across years. All sites appear to have regular recruitment with no gaps in size ranges across the two years.

In Region 2 the size frequency histograms for *P. vulgata* show large differences between wooden and rock groynes in both 2015 and 2016 (Figure 3.9). The rock groynes had a larger abundance of limpets and a size range of 2-60 mm whereas the wooden groynes had a lower abundance and a size range of 14-60 mm as there was a lack of young individuals on the wooden groynes. The natural shore at Shanklin shows consistent recruitment and growth unlike the natural shore at Bembridge which shows a large reduction in smaller sized limpets from 2015 to 2016 (Figure 3.9). Overall the rock groynes show the most similar recruitment pattern to the natural shore than the wooden groynes.

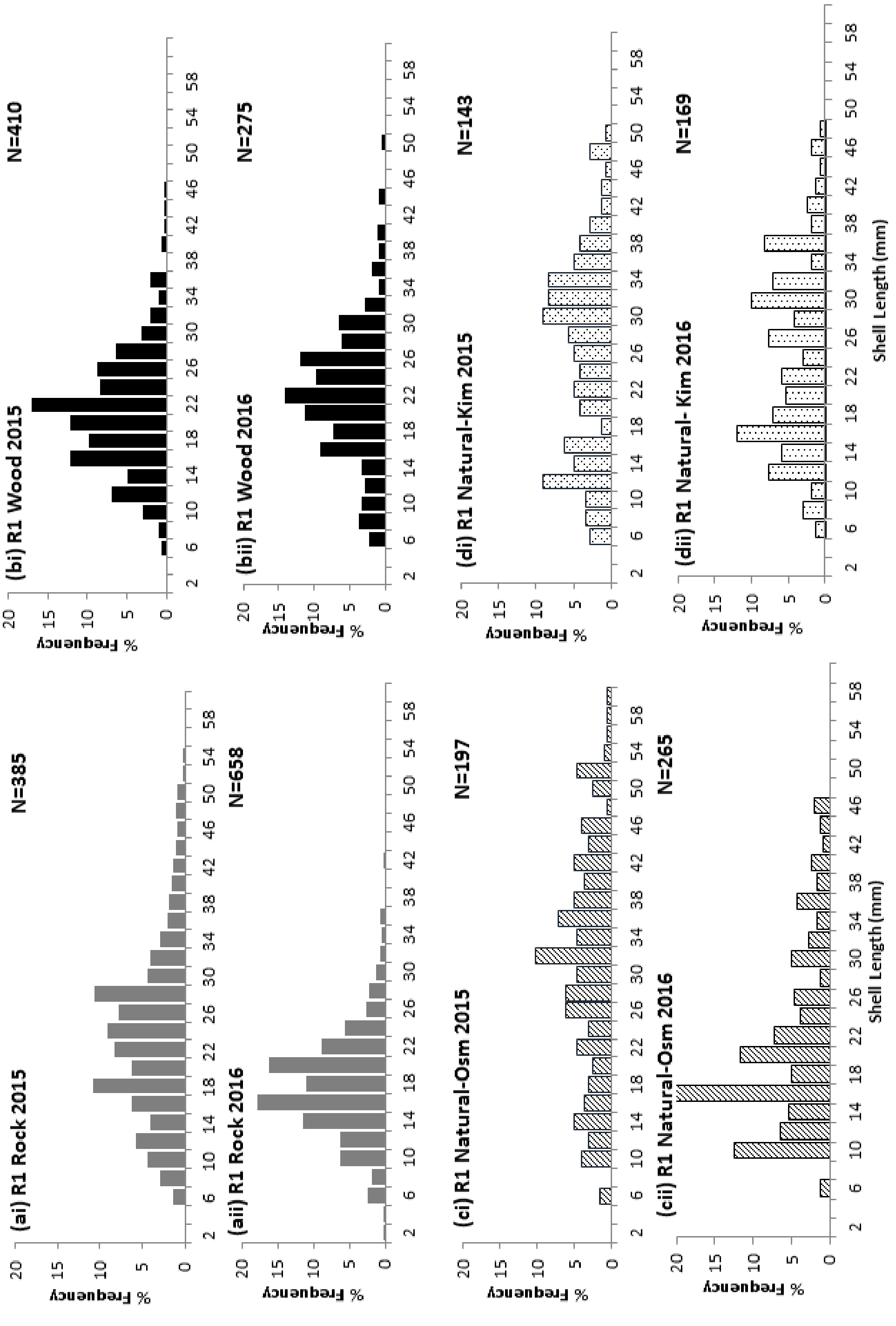

Figure 3.8 Size frequency histogram for *Patella vulgata* shell length (mm) for Region 1 on Bournemouth (a) rock groynes and (b) wooden groynes and natural shores (c) Osmington Mills and (d) Kimmeridge during (i) 2015 and (ii) 2016 (N= number sampled).

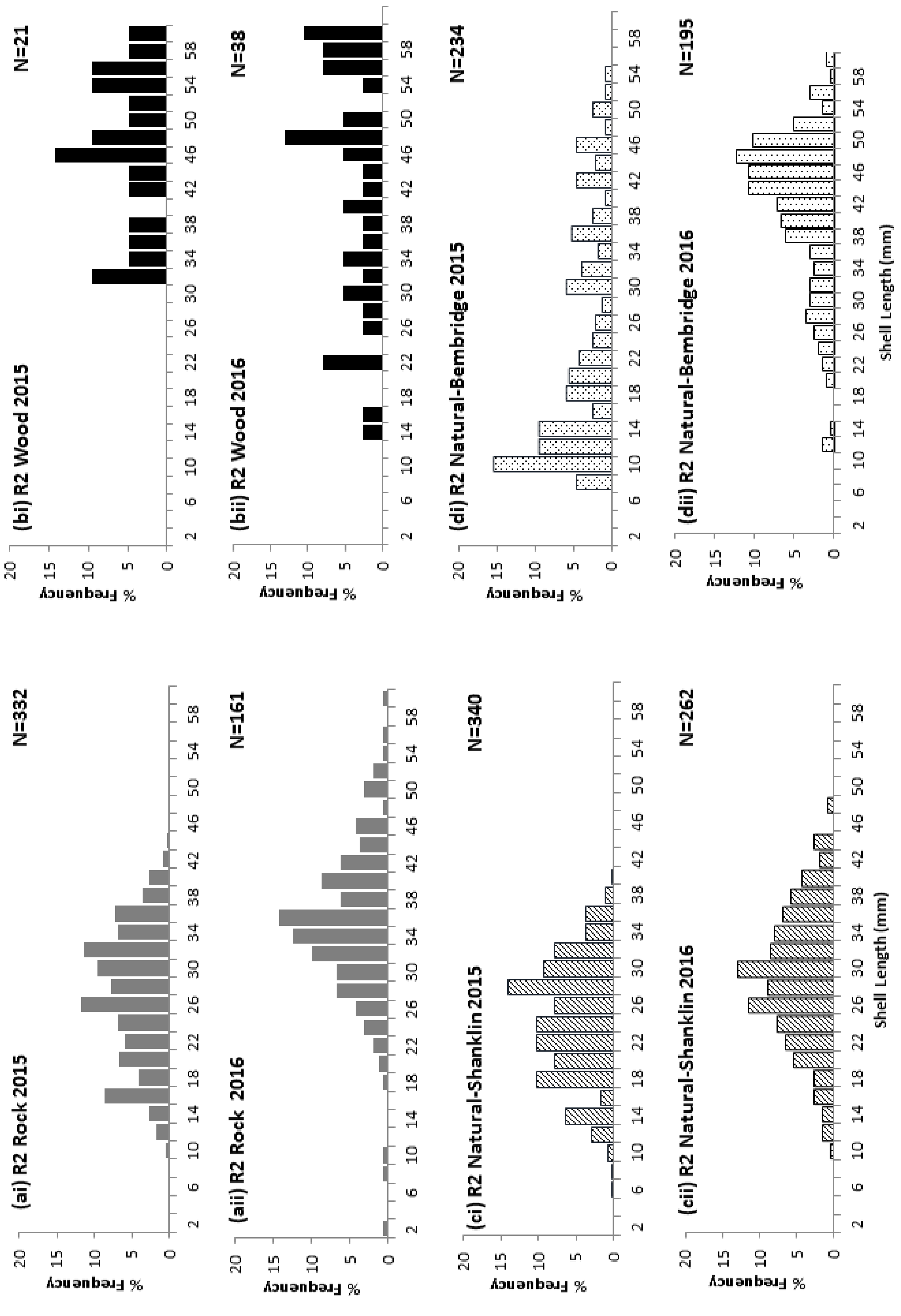

Figure 3.9 Size frequency histogram for *Patella vulgata* length for Region 2 on Lee-on-Solent (a) rock groynes and (b) wooden groynes and natural shores (c) Shanklin and (d) Bembridge (i) 2015 and (ii) 2016 (N= number sampled).

3.3.3 Mussel recruitment and survival

In 2015 the *M. edulis* on the rock groynes ranged from 4-44 mm in size, whereas the individuals on the wooden groynes ranged from 4-52 mm (Figure 3.10) with fewer mussels of size 16-28 mm. In 2016 the mussel populations were absent from the rock groynes at Bournemouth and the wooden groynes had an extremely high number of new recruits (2-6 mm) and older individuals (40-52 mm) but a lower number of mid-sized mussels (6-30 mm). The Calshot samples from the natural population showed two separate age classes, with a higher abundance of new recruits (4-6 mm) in 2016 (Figure 3.10).

In Region 2 the size ranges for *M .edulis* on wooden an rock groynes were similar however there was a larger proportion of younger specimens (0-6 mm) on the rock groynes, particularly in 2016 (Figure 3.11), whereas the wooden groynes had a larger proportion of mid to large sized mussels (22-34 mm). Overall compared with the natural shore at Calshot there were significantly higher abundances of mussels on the artificial wooden and rock groynes than the natural shore (Figure 3.11).

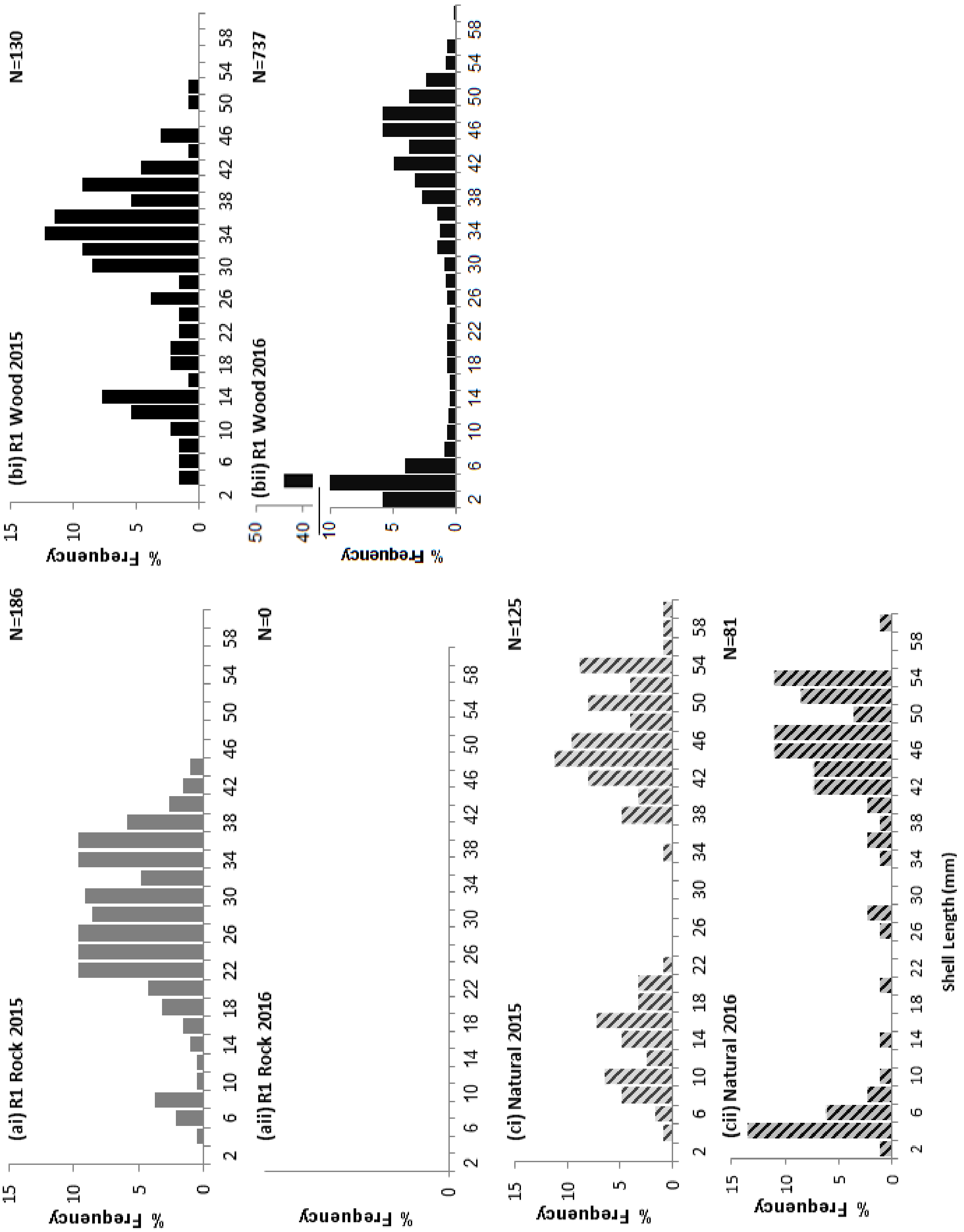

Figure 3.10 Size frequency histogram for *Mytilus edulis* shell length (mm) for Region 1 on Bournemouth (a) rock groynes and (b) wooden groynes and natural control site (c) Calshot during (i) 2015 and (ii) 2016 (N= number sampled, note variation in axis for bii).

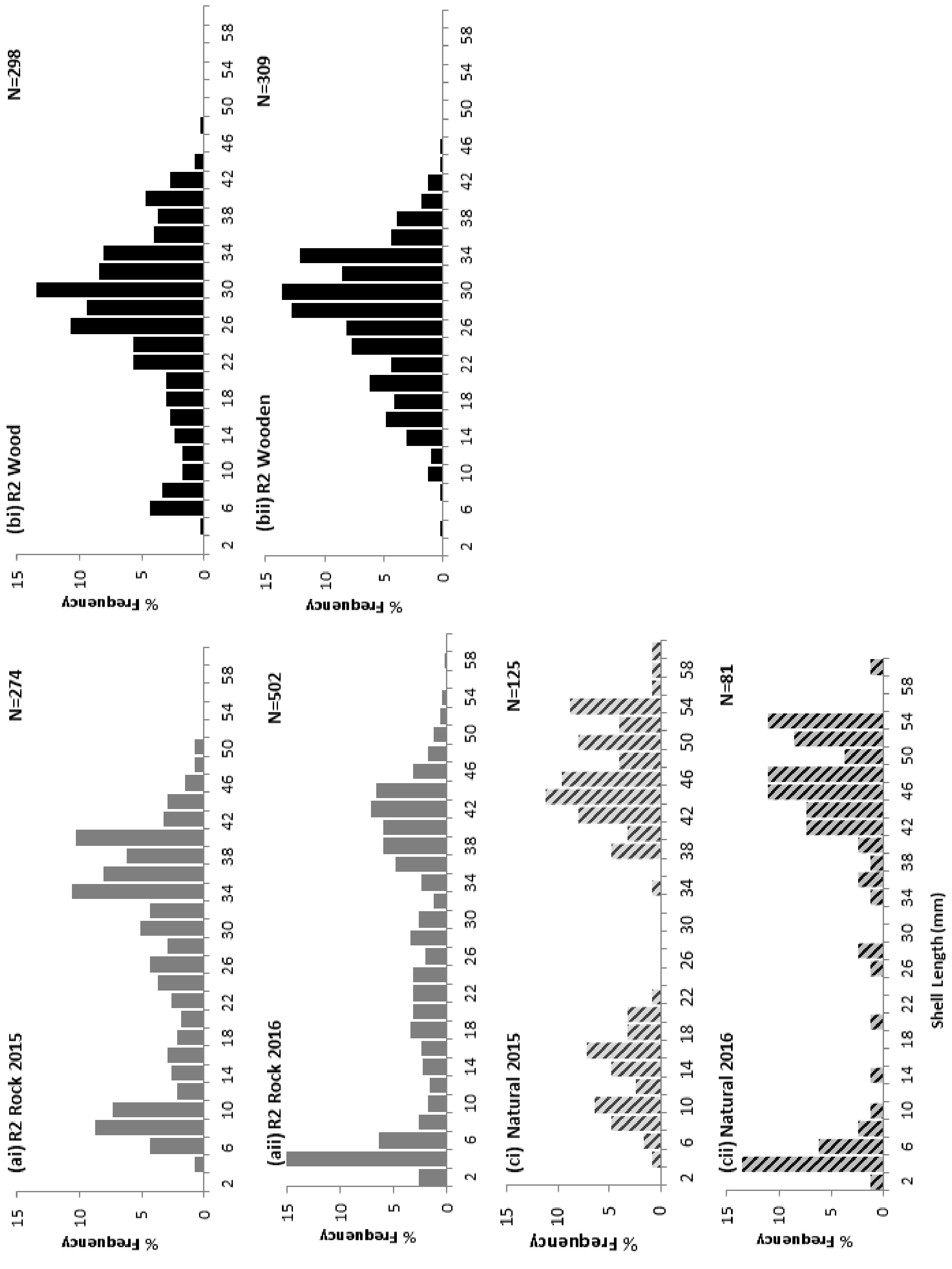

Figure 3.11 Size frequency histograms for *Mytilus edulis* shell length (mm) for Region 2 on Lee-on-Solent (a) rock groynes and (b) wooden groynes and natural control site (c) Calshot (i) 2015 and (ii) 2016 (N= number sampled).

3.4 Discussion

A comparison of the settlement, recruitment and mortality of *S. balanoides, P. vulgata* and *M. edulis* on wooden and rock groynes and natural shores has to date never been investigated. This study found that both the wooden and rock groynes supported populations of all three species, however species-specific variation of substrate preference was observed for recruitment and mortality across regions. Understanding the processes that affect the colonisation of artificial structures, particularly in relation to substrate type, will allow us to predict the potential community development and determine if the communities present on the structures are stable.

S. balanoides, settlement, recruitment and mortality

The recruitment of *S. balanoides* was greater on artificial structures in comparison to natural shores across both regions; the wooden groynes supported the greatest abundance in Region 1 whereas the rock groynes supposed high numbers of recruits within Region 2. The variation in wooden and rock groyne recruitment appears to be site-specific although Anderson & Underwood (1994) has previously reported a greater number of barnacles on plywood panels compared with concrete and fibreglass and aluminium and fibrous/porous substrata are known to be more attractive than hard smooth surfaces (Pomerat & Weiss, 1946). The inconsistency between regions could be attributed to the wooden groynes in Region 2 being deteriorated (soft and flaky) during the study period and poorly maintained, resulting in a potentially less favourable/ stable habitat. Herbert & Hawkins (2006) found that the density of the barnacle *Chthamalus montagui* on soft flaky substrate prone to erosion, such as chalk, is low due to high rates of post recruitment mortality. As rock groynes provide a more physically similar habitat to natural shores in terms of substrate hardness, this may explain the preferential settlement on rock groynes at particular sites. In addition increased surface texture on rock groynes will reduce desiccation stresses (Firth et al., 2013b) which provides increased protection from predators (McGuinness &

Underwood, 1986) and create a greater surface area for settlement (Anderson & Underwood, 1994). These features should be considered and incorporated into new infrastructure designs in order to encourage settlement and survival of intertidal species.

Overall there was found to be greater post settlement mortality on the artificial substrates compared to the natural shores. This could primarily be explained by the variation in wave action (Vadas et al., 1990) as the groynes experience higher levels of wave energy due to their orientation (Perdok, 2002; Harlow, 2013). Yet aspect of the artificial structures was not found to be a significant factor in the post settlement mortality of *S. balanoides*, as found by Pinn et al (2005). Wooden groynes had significantly higher rates of mortality than rock groynes which could be attributed to substrate type. The substrate type will influence the desiccation stress experienced by cyprids, metamorphs and new recruits, as the hardness, thermal capacity (Raimondi, 1988) and surface heterogeneity (Herbert & Hawkins, 2006; Conlan, 2013) will vary between substrates and sites depending on the age of the structure (Raimondi, 1988; Pinn et al., 2005). Biological processes such as competition and predation (Connell, 1961a; Gosselin & Qian, 1996) can also influence the recruitment levels on natural and artificial substrates. Although quadrats were cleared prior to this experiment, the wooden groynes in Region 2 were quickly recolonised by ephemeral algae (*Ulva* spp.) during both monitoring years. As algal sweeping is known to reduce the survival of barnacle cyprids (Jenkins et al., 1999c) it is probable that the algal growth resulted in the high post settlement mortality of *S. balanoides* within Region 2. Although no direct comparison was made between substrates, the rock groynes had greater abundances of limpets, therefore there is a greater potential for limpet bulldozing which will contribute to the post settlement mortality of barnacles (Hawkins, 1983). The influence of predation is thought to be minimal in this study as either none or very few predators were observed in the experimental plots or in the adjacent vicinity, although it is possible that predation from fish and crabs occurred when the groynes were submerged (Burrows et al., 1999; Mauck & Harkless, 2001). Natural processes such as density dependant mortality could have also been a factor which contributed to the variation in mortality rates in the study (Holm, 1990; Knights & Walters, 2010). The wooden groynes provided a smaller surface area of space to colonise resulting in higher densities of "standing stock" barnacles present compared to the natural shores and rock groynes. Rock groynes provide a much larger surface area in which to colonise, resulting in the "standing stock" being spread over a larger area in lower densities.

P. vulgata recruitment and mortality

The abundance of *P. vulgata* was greater on the artificial substrates in comparison to the natural shores, yet all sites exhibited stable recruitment with the exception of the wooden groynes in Region 2 which showed no recent recruitment. The lack of recent recruitment could be attributed to the poor quality habitat provided by the deteriorated wooden groynes and the dense coverage of *Ulva* spp which colonised after clearances were conducted. The rock groynes revealed recruitment patterns most comparable to the natural shores which could be attributed to them having similar surface characteristics and chemical compositions to the natural shores. In comparison, the wooden groynes provide a novel substrate which is not commonly found in the marine environment in temperate regions. However the age of the structure can affect the amount of weathering which has occurred on a substrate (Coombes et al., 2011), varying the surface texture, as observed with the deteriorated wooden groynes within Region 2. In addition, age has been shown to effect the assemblage composition on artificial structures, with new groynes (<2 years old) being dominated by ephemeral species and older groynes (6+ years) being dominated by perennial species (Pinn et al., 2005). This study found that once a groyne had reached an age (40 years +) at which it was deteriorated and providing poor quality habitat the community then resorted back to ephemeral species such as *Ulva sp.*. Implying that if structures are not regularly maintained and do not provide adequate habitat in isolated areas which lack natural rocky shores, then the metapopulation of a region could be effected.

M. edulis recruitment and mortality

Natural intertidal mussel beds are relatively scarce in both regions, although populations exist on the mixed cobble and sediment shores at Calshot. Lee on Solent had greater abundances of mussels on both the wooden and rock groynes in comparison with the natural shore, implying that mussel populations were more stable and have a preference for artificial structures in this region. Primary mussel settlement is associated with filamentous algae (Seed, 1969; Bayne, 1976), which were most abundant on the artificial structures in particular the wooden groynes. Mussels are also known to favour rough surfaces in which turbulent flows are created, therefore the dynamic water movement around the groyne structures may provide suitable conditions to facilitate filter feeding, which could explain the consistent growth rates of *M. edulis* on the majority of the groynes (Holt et al., 1998). The absence of *M. edulis* on the rock groynes in Region 1 during 2016 was attributed to the sand nourishment scheme which smothered the groynes that the mussels occupied. Moschella et al., (2005) found that structures built in soft sediment habitats are prone to sand scouring and smothering, which can have a negative impact on community establishment as observed here. This highlights the effect in which anthropogenic disturbances can halt or reset the community succession on coastal defence structures (Airoldi & Bulleri, 2011).

Ideally research on settlement and recruitment patterns should be conducted over multiple years to determine the long term fluctuations in larval supply. However due to time restraints this was not possible for the current study. It would also be useful to quantify the amount of barnacle larvae available in the water column during the settlement period at each site in order to obtain an accurate settlement rate. One drawback of this study was that the effect of substratum could not be separated from the effect of structure; in order to test this, wooden panels would need to be attached to rock groynes and vice versa. Yet due to the prevalence of the wood boring crustacean gribble (*Limnoria* spp.) in this region, no holes/attachments were permitted to be drilled into the wooden groynes. Future work should focus on the long term fluctuation in settlement and recruitment of communities on artificial structures, particularly in relation to the influence of artificial substratum and human disturbances such as maintenance activities on artificial structures.

Conclusions

Overall fluctuations in populations of barnacle, limpets and mussels on artificial structures did not differ significantly to those on natural shores, with the exception of high mortality on the deteriorated wooden groynes in Region 2. Across both regions the artificial structures were observed to be favourable for settlement and recruitment of *S. balanoides* and *M. edulis*. Regional variation in substrate preference was observed in the settlement of *S. balanoides,* however with the exception of the wooden groynes in Region 2, *P. vulgata* and *M. edulis* populations were stable across all substrates. The mortality of *S. balanoides* was higher on the wooden groynes in comparison to the rock groynes, yet overall aspect/exposure did not have an impact on mortality rates of *S. balanoides* on either wooden or rock groynes. If natural populations were to suffer a disturbance event they would have the potential to be repopulated by larvae from the artificial structures, depending on suitable hydrodynamic processes (Levins, 1969). The creation of artificial habitats also enables the exchange of genetic diversity and the potential dispersal of species beyond their natural range. Notwithstanding human disturbances, species populations on groynes may be sufficiently stable in some regions and contribute to metapopulations. Providing structures do not physically deteriorate processes involved in structuring the communities on artificial structures are regionally comparable to those on natural shores.

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4 The Influence of Piers on Surrounding Communities of Mobile Fauna

Abstract

The influence of coastal artificial structures on surrounding benthic and mobile communities has rarely been studied. Here, patterns of species richness and assemblage composition of mobile and benthic communities were investigated around two piers on the south coast of England. Baited Remote Underwater Video revealed that there was a higher species richness of fish at the stations closest to the pier and at the outer (seaward) end of the pier. Benthic infauna in sediments revealed no significant distribution patterns in relation to the pier. Stable Isotope Analysis of representative species within surrounding communities showed that the pier pilings, which are colonised by dense epifauna and flora, could provide foraging opportunities and shelter for small fish such as smelt (*Osmerus eperlanus*), which in turn provides an abundant food source for larger species such as bass *(Dicentrarchus labrax)* and pouting *(Trisopterus luscus*). The pier could therefore provide both shelter and a foraging area for multiple species at varying trophic levels and enable connectivity to intertidal habitats. Understanding the effects artificial structure have on multiple trophic levels of marine species enables us to assess the connectivity and wider implications of constructing additional structures in the marine environment.

4.1 Introduction

Increased construction within the marine environment is inevitable due to population increases and industrial activities. Hard artificial structures such as offshore windfarms, oil rigs, artificial reefs and marinas are being constructed across the globe, when they are located in areas of soft sediment they change the habitat composition for marine species in turn altering the ecosystem. The addition of hard artificial substrata in most soft sediment environments increases opportunities for the colonisation by species which were previously absent due to the lack of available structural habitat (Airoldi et al., 2005; Cenci et al., 2011). As this is true for both native (Johannesson & Warmoes, 1990) and non-native species (Bulleri & Airoldi, 2005), investigating the connectivity between structures and natural habitats is crucial for predicting the spread or expansion of populations (Herbert et al., 2003; Cenci et al., 2011; Rius et al., 2014). Understanding how artificial structures influence the trophic dynamics within communities is also important to gain an ecological awareness of their wider impacts (Aguilera, 2016). Any artificial structure added to the marine environment has the potential to become an artificial reef is colonised by marine organisms (Rilov & Benayahu, 1998; Wilhelmsson et al., 2006) particularly in areas which lack natural reefs/habitats.

Artificial reefs are used to increase fish stocks around the world (Carr & Hixon, 1997; Grossman et al., 1997; Keller et al., 2017) by providing food and refuge to fish in areas of limited habitat (Carr & Hixon, 1997). There has been huge debate as to whether artificial reefs work as an attractant or a producer of fish biomass (Bohnsack & Sutherland, 1985; Seaman, 1996; Pickering & Whitmarsh, 1997). Some structures may act as an aggregation device and attract species from adjacent habitats (Bohnsack, 1989) whereas some may create additional habitat which can provide shelter, foraging opportunities and recruitment space to in turn increase the carrying capacity and fish biomass (Bortone et al., 1994). The design of the artificial reef is important in maximising the potential for structures to become producers rather than attracters

(Pickering & Whitmarsh, 1997). In addition to the protection provided from predation, the benthic invertebrates and algae attached to the structure provide a vital food resource for mobile species such as fish and crustaceans (Lindquist & Cahoon, 1994; Cresson et al., 2014).

Connectivity

Artificial structures may alter the metapopulation of a region by altering the connectivity between isolated habitats and/or populations (Vega Fernández et al., 2008). The majority of pelagic fish species travel relatively large distances, using habitats as stepping stones for foraging and shelter (Vega Fernández et al., 2008). For resident fish species, the lack of suitable foraging areas in close proximity is known to reduce the density of fish populations (Caddy, 2007). Territorial or benthic fish such as gobies (*Gobiidae* spp.) have a higher dependency on well-connected habitats due to their increased predation risk when crossing sandy areas (Walsh, 1985; Belmaker et al., 2005). Studies in Sicily (Mediterranean Sea) have shown the addition of artificial structures increases connectivity, resulting in higher fish species richness and abundance (Vega Fernández et al. 2008). Small reef fish tend to avoid crossing bare areas of sand from tens to hundreds of metres wide (Frederick, 1997; Chapman & Kramer, 2000; Nanami & Nishihira, 2003), creating potential barriers to species movement. Thus, the presence of artificial structures potentially encourages these fish to move greater distances due to the increased refugia provided from predators and the environmental conditions.

Piers

Piers are a common feature on urbanised coastlines, yet little is known about their influence or impact on marine communities. Piers are used for recreation, tourism and angling, with the additional function of berthing large pleasure boats. They are constructed perpendicular to the shore and extend from the intertidal zone to the sublittoral. The size and structural complexity of piers differ; some are constructed with simple upright piles whereas others have more complex interlocking cross members. Construction materials vary depending on location and purpose but are normally composed of traditional materials such as metal, wood and concrete. Piers have the potential to become fish aggregation devices (FADs) and / or secondary artificial reefs (Rilov & Benayahu, 1998; Wilhelmsson et al., 2006) particularly in areas which lack natural reefs/ habitats. As with artificial reefs the colonisation of epifauna on pier pilings will in turn attract mobile species such as fish and crustaceans and in time increase the biomass of the habitat. Research from Australia has predominantly studied the epifaunal communities attached to the piers and revealed dominance by ascidians, sponges and encrusting bryozoans (Kay & Butler, 1983; Glasby, 1999a). High prey availability on artificial structures has shown to attract fish and mobile invertebrates (Pickering & Whitmarsh, 1997; Reubens et al., 2011, 2013; Krone et al., 2013).

Mobile fauna

The majority of research on the association of mobile fauna with pier pilings has been conducted in estuarine conditions. In Sydney Harbour, Coleman and Connell (2006) found a correlation between the abundance of fish and the amount of epibiota on the pilings, but only for one species *(Trachinops taeniatus).* They concluded that mobile species associated with artificial structures feed in adjacent habitats as with artificial reefs, which has been supported by stomach content analysis (Randall, 1963; Lindquist & Cahoon, 1994). Coleman & Connell (2006) also found that shelter provided by the pier piling is relative to the size of the fish (Hixon & Beets, 1989), therefore epifauna (such as algae) may be successful at creating shelter for smaller bodied fish, but less so for larger fish species. The authors also infer that fish respond to the presence or absence of pier pilings on a greater scale than the presence of the pier epifauna alone (Hixon & Beets, 1989). Fish are known to aggregate around prominent changes in topography, with Randall (1963) reporting a higher abundance of fish associated with taller structures on artificial reefs. The depth of water in which the pier is located will affect the community composition of both epifauna and mobile species, as particular species inhabit certain water depths (Bell, 1983; McGehee, 1994; Connell, 2001; Harvey et al., 2007; Herbert et al., 2017). Water depth affects light levels, which have shown to also have a strong impact on subtidal epifaunal assemblages (Glasby, 1999b; Saunders & Connell, 2001). Studies have suggested that certain visual feeders, such as salmon *(Oncorhynchus* spp.) and winter flounder (*Pseudopleuronectes americanus*)*,* avoid using the shaded areas underneath piers and instead utilise the lit areas adjacent to the pier (Able et al., 2013; Munsch et al., 2014). However scavengers such as flatfish and crustaceans are known to be attracted to and forage upon the build-up of biological detritus at the base of artificial structures (Boehlert & Gill, 2008; Coates et al., 2011).

Facultative piscivorous fishes such as bass *(Dicentrarchus labrax)* are particularly known for using the shelter provided by piers on stretches of featureless coasts and can position themselves in slack water behind piers and jetties in order to catch the smaller prey species which may be swept past by the current (Pickett & Pawson, 1994). This is similar to planktivorous fish that can also use the currents created by vertical structures to feed on plankton washed past (Rilov & Benayahu, 1998).

Surveying mobile species around structures

Various techniques can be used to quantify subtidal mobile populations associated with artificial structures, the majority of which have been previously used to survey artificial reefs. Destructive methods, such as seine netting, trawling and long line fishing, can be used to quantify fish abundance (Newman et al., 1997; Mahon & Hunte, 2001; Soldal et al., 2002; Cappo et al., 2004). However, these techniques are not ideal around structures such as piers due to the danger of entanglement on the structure. Non-destructive methods, such as hydro-acoustic surveys and observational surveys, are preferred in sensitive and protected areas (Malcolm et al., 2007). Hydro-acoustic survey methods can be used to estimate fish abundances but interpretation of the results can be limited due to taxonomic ambiguity (Simmonds & MacLennan, 2005). A more common method is observational studies by snorkelers and scuba divers which can be used to estimate species abundance with greater accuracy (Coleman & Connell, 2006). There are, however, various limitations of snorkel/diver surveys, including the presence of the diver/snorkeler deterring species, water visibility impacting the consistency of data, the limitation imposed by water depth (Harvey et al., 2013; Harasti et al., 2015), and cryptic species often being underrepresented in the data (Ackerman & Bellwood, 2000; Clynick et al., 2007; Brandl et al., 2017). In addition, the currents flowing around artificial structures can be dangerous for divers particularly in high energy nearshore environments.

Baited Underwater Video (BUV), or Baited Remote Underwater Video (BRUV), is an established methodology for calculating biodiversity and monitoring fish populations in both marine and estuarine habitats in Australia (Cappo et al., 2004; Stobart et al., 2007; Harvey et al., 2012; Lowry et al., 2012; Folpp et al., 2013). BRUV is a non-destructive sampling technique favoured over more destructive methods such as trawling, making it an ideal way of surveying marine protected areas (MPAs) and no-take zones (Gladstone et al., 2012; Sheehan et al., 2016). BRUV can be used in a variety of circumstances, including environments inaccessible to SCUBA divers and snorkelers (Willis et al., 2000) including artificial reefs (Herbert et al., 2017). Yet to the best of current knowledge, the technique has not previously been employed to investigate the fauna associated with coastal piers or the sphere of influence of structures. Various studies have compared the effectiveness of BRUV in contrast to underwater visual census (UVC) (Colton & Swearer, 2010; Lowry et al., 2012), trawl sampling (Cappo et al., 2004) and angling (Willis et al., 2000). Whilst BRUV has advantages in being non-destructive, concerns have been raised about the quality of results in turbid conditions, alongside poor representation of cryptic species and the bias towards mobile predators. Optimal deployment time has received much debate recently, with some studies suggesting 90 minute deployments (Gladstone et al., 2012) and others recommending 20 minutes (Stobart et al., 2007). The type and amount of bait has been shown to influence experimental outcomes (Dorman et al., 2012; Hardinge et al., 2013; Wraith et al., 2013), as variation in assemblages, fish size and bait depletion have been recorded (Harvey et al., 2007; Dorman et al., 2012; Hardinge et al., 2013; Wraith et al., 2013). Studies suggest that the number of predators/ scavengers could be overestimated using bait (Harvey et al., 2007) or attract particularly large fish (Hardinge et al., 2013). Yet Harvey et al., (2007) found that bait provided a better representation of individual species within an assemblage when compared with non-baited video. Evidence has also shown that there may be species specific variation in the attraction to different bait types (Dorman et al., 2012). However, oily fish such as sardines and mackerel, have shown to produce consistent results (Heagney et al., 2007; Martinez et al., 2011; Dorman et al., 2012; Wraith et al., 2013; Schmid et al., 2017).

Investigating food webs

Stomach content analysis has traditionally been used to investigate food web dynamics (Hyslop, 1980; Lindquist & Cahoon, 1994; Baker et al., 2014). However, it only provides a short term representation of the diet, resulting in samples having to be collected over time and space in order to understand foraging and dietary patterns (Vizzini & Mazzola, 2002). More recently, developments in stable isotope analysis *(*SIA) have increasingly enabled the longer term diets of consumer species to be estimated using the stable isotopes of carbon (^{13}C) and nitrogen (^{15}N). The ratios of ^{12}C:^{13}C enable the energy sources of consumers to be predicted from their putative food resources, whilst the ratios of ^{14}N:^{15}N enable the reconstruction of the trophic structure of the food web (Fry et al., 1999; Post, 2002). This is because there tends to be a step-wise enrichment of the stable isotopes up the food chain, enabling the trophic relationship between species and food resources to be identified via the application of fractionation factors between the consumers and their putative prey (Vander Zanden & Rasmussen, 1999; Post, 2002). These data now also enable metrics such as trophic niche size of each species to be determined (as isotopic niche size) (Jackson et al., 2011, 2012), with a number of mixing models now developed that allow diet composition to be predicted for a consumer from the stable isotope data of their prey (Phillips et al., 2014).

To obtain reliable estimates of diet composition of a consumer requires an accurate estimation of the discrimination factor (also known as fractionation/enrichment factor), the step wise enrichment of isotopes from food to consumer (Bond & Diamond, 2011). Whilst a large amount of research effort has been expended on this, there remains some uncertainty for specific species over the most appropriate value to use, with considerable differences often evident between herbivorous and piscivorous diets (Phillips et al. 2014). In addition, the isotopic turnover rate (the time taken for the tissues of the consumer to reflect the isotopic composition of the food resource) is also important to help understand the variation in different tissue turnover rates (Buchheister & Latour, 2010; Ankjærø et al., 2012). Previous studies have shown variable discrimination and turnover rates for fish, highlighting a need for species-specific data for accurate predictions (Bond & Diamond, 2011). In addition, there are also considerable differences in discrimination and turnover rates between tissues of the same species, with more metabolically active tissues tending to have higher turnover rates (Vander Zanden et al., 2015). In this regard, blood tends to have a relatively short half-life, whereas muscle, fin tissue and scales tend to have longer half-lives (Vander Zanden et al., 2015).

To date, few studies have determined the extent to which artificial structures are used for foraging (Metzger et al., 2001; Coleman & Connell, 2006; Able et al., 2013). The use of SIA and SCA will allow us to recognise the spatial and temporal diet composition of fish around the piers, in order for us to determine if fish use pier as foraging areas.

Given the scarcity of data on how piers influence local patterns of diversity and the trophic dynamics of species, the aims of this chapter are to: a) Determine how pier structure influences the distribution of mobile fauna between intertidal and subtidal waters; b) Quantify whether mobile fauna are found in higher abundances closest to the pier compared with increasing distance away from the pier; c) Identify if mobile fauna utilise the pier structures as foraging areas.

The hypotheses tested are:

(1) The species richness and species abundance of mobile fauna will decrease with distance away from the pier (Figure 4.1, Model A).

(2) The species richness and species abundance of mobile fauna will increase with distance along the pier i.e. from shallow to deep water (Figure 4.1, Model B).

(3) The species richness and total abundance of benthic infauna will decrease with distance away from the pier (Figure 4.1, Model, A).

(4) The species richness and total abundance of benthic infauna will decrease with distance along the pier (Figure 4.1, Model B).

(5) Fish are utilising both the epifauna on the pier pilings and the smaller fish attracted to the pier as a food resource.

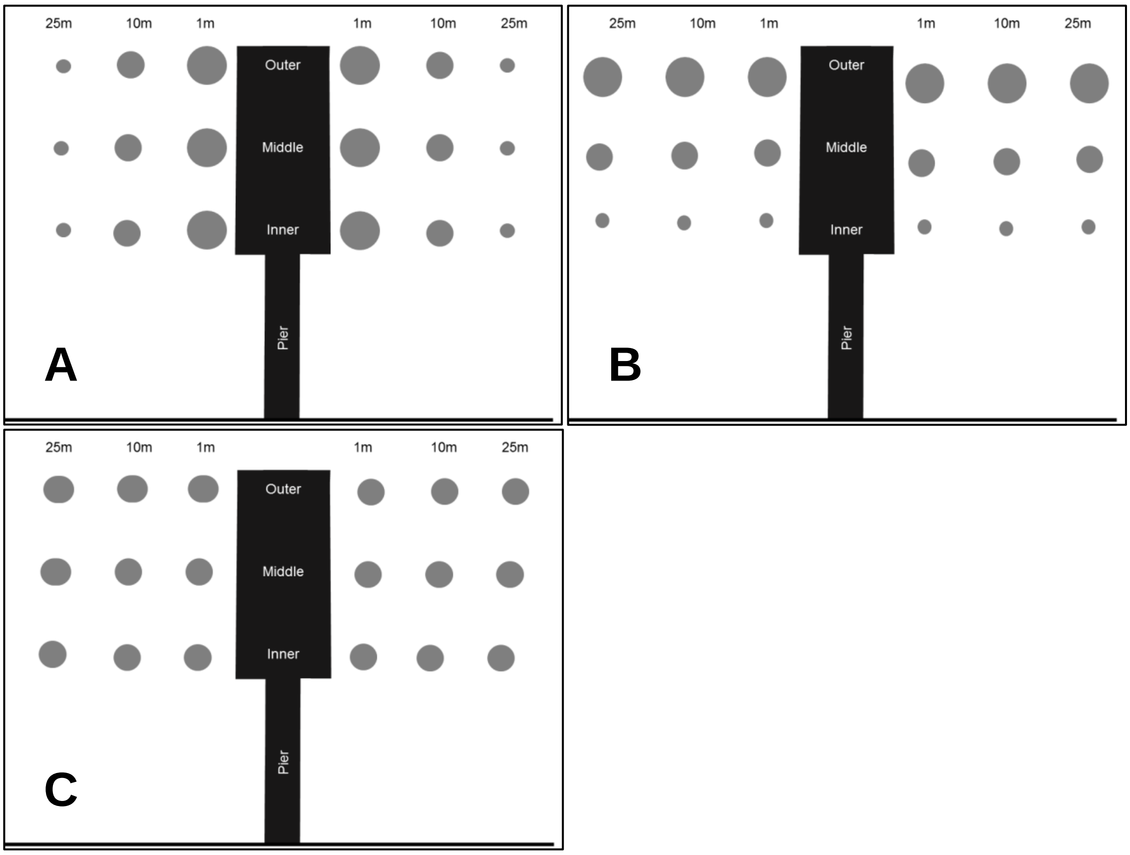


Figure 4.1 Diagram to represent models of species distribution around the pier structure (in black) and the varying distances along the pier (Inner, Middle, Outer) and distances away from the pier (1m, 10m, 25m). Model A: Higher abundance and richness closest to the pier, Model B: Higher abundance at outer stations, Model C: No difference in abundance. (Size of grey circle relate to abundance/richness of fauna, large circle= high, medium circle= intermediate, small circle= low)

4.2 Methods

4.2.1 Study Sites

Two piers which broadly represent typical pleasure piers encountered around the British coast were selected on the south coast of England: Bournemouth Pier (50.7140 ^{o} N -1.8745 ^{o} W) and Boscombe Pier (50.7181^{o} N -1.8428^{o} W). These piers which are predominately used for tourism, recreation and angling, are located within Poole Bay, which is a moderately exposed bay with prevailing south-westerly winds and a tidal range of 1.7 m during spring tides and 1.3 m during neap tides. The seabed within the vicinity of the piers is mostly medium sand, with increasing mobility inshore. The closest natural reef (7 m deep) is located approximately 850 m south west of Bournemouth Pier and Boscombe Artificial Surf Reef (5 m deep) is located 210 m east of Boscombe Pier. Bournemouth Pier is situated 2.29 km west of Boscombe Pier and differs in material substrate type and structural complexity (Table 4.1). The pilings at Bournemouth Pier are constructed in a complex manner with various interlocking joints and cross sections. In comparison, Boscombe Pier is 90m shorter and structurally simpler with a much smaller submerged surface area (Table 4.1). Both piers were constructed in the 1800s and have had various degrees of renovation and pile replacement. Natural subtidal reefs are uncommon in the study region and those present are isolated and surrounded by soft sediment. The addition of artificial structures has increased the potential connectivity of patches between both natural and artificial habitats.

Table 4.1 Comparison of Bournemouth and Boscombe Piers in terms of age, materials and size. Neck width refers to the section attached to the shore. Head width refers to the wider seaward, outer end of the pier.

Pier	**Build date**	**Materials**	**Length (m)**	**Width at Neck (m)**	**Width at Head (m)**	**Height above seabed (m)**	**Water depth at end (m)**	**Submerged surface area (m^2)**
Bournemouth	1856 Reconstructed in 1979-81	Cast iron & concrete with wooden landing stages	270	12	40	5.79	~ 4	2797
Boscombe	1888 Reconstructed 1958-60	Concrete	180	10	16	5.79	~ 3	320

4.2.2 Survey methods

Mobile communities

Mobile communities were surveyed using Baited Remote Underwater Video (BRUV). The BRUV device consisted of a GoPro® Hero 2 or 3 video camera in a waterproof housing mounted onto a large aluminium frame which was then deployed over the side of a boat or pier. The metal frame is 0.5 m square, 0.7 m tall and weighs 11.5 kg, the frame has a rope and 30cm diameter buoy attached which allows for the unit to be relocated from the surface (Figure A2.2). A 5mm mesh bait basket fixed to the end of a 1 m long plastic bait arm protruded in front of the cameras field of view. For this study 100 g of chopped fresh mackerel was used as bait, as in previous studies (Martinez et al., 2011) and the pilot study showed oily fish to be more attractive to a wide range of mobile fauna compared to ragworms and pelletized fishmeal.

Between June-September 2015, BRUV deployments were carried out at 18 sampling sites around each pier (Figure 4.2). These were located at three fixed distances along the pier (Outer, Middle, Inner) and three distances away from the pier (1 m, 10 m, 25 m) on both the east and west sides (Figure 4.2). The inner distance for both piers corresponded to Extreme Low Water Springs (ELWS) and distances away from the pier were measured from the boat using a Nikon Laser 1200 Long Range Rangefinder. Deployments were conducted from a small boat which departed the area once the BRUV was deployed. These locations were chosen following several pilot studies. They initially included distances of 50 m away from the pier, but as the pilot revealed no difference in abundance and species richness at 50 m compared to that recorded at the 25 m, the 50 m sample was considered uneeded due to time and financial restraints and not included in subsequent sampling. Each sampling station for each pier was replicated three times during the sampling season. Each deployment consisted of the BRUV unit being placed in the water for 20 minutes allowing 5 minutes for conditions to settle, after this time the BRUV unit was brought back to the surface.

Pilot studies comparing deployments of 15, 30 and 40 min duration (Figure A2.3) revealed that 20 minutes was adequate to obtain reliable measures of abundance and species richness.

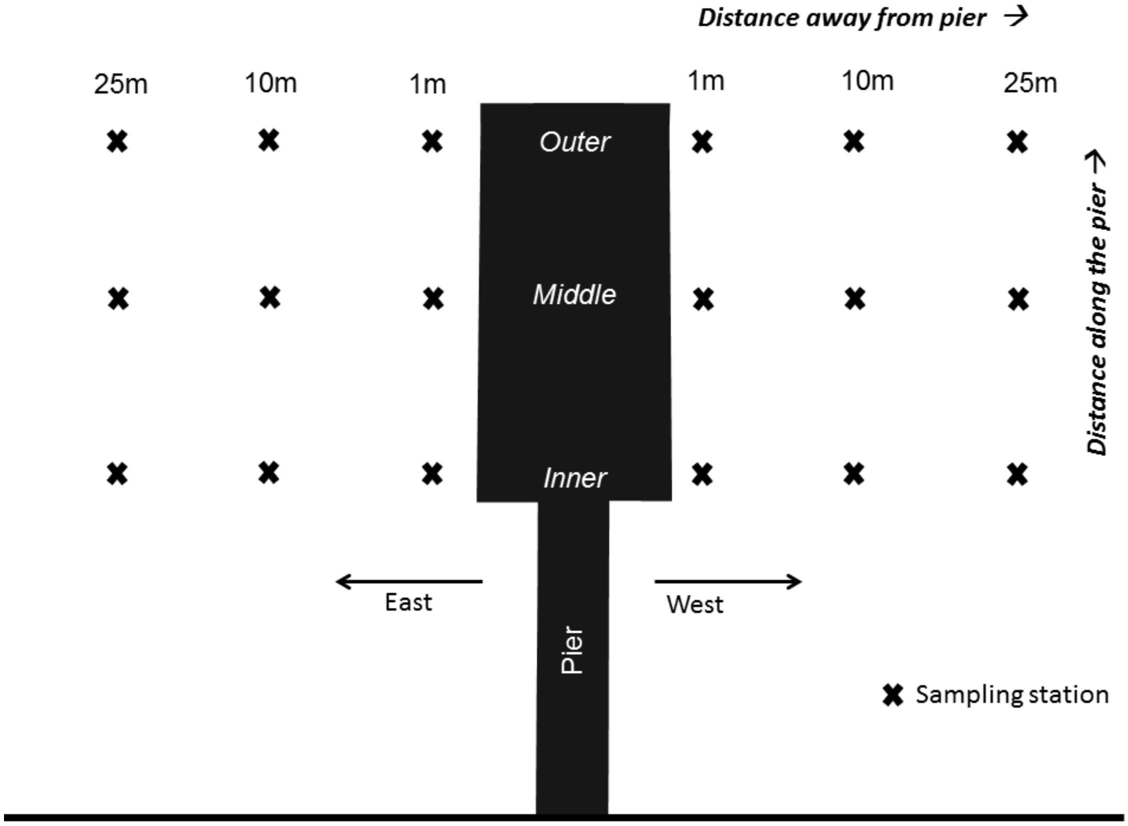


Figure 4.2 Sampling design of the pier BRUV deployment stations.

Epifaunal communities on pier pilings

In order to determine the distribution of potential food available from the fouling on the pier pilings, underwater video cameras (GoPro® Hero 3) were attached to the end of a 2 m pole and deployed over the side of a small boat. At each pier, six vertical video transects were recorded down the length of the outer faces of six pier pilings, two piles were surveyed within each section (Inner, Middle, Outer). The videos were analysed using Media Player to determine the abundance and coverage of algae and main functional groups of epifauna. These abundances were summarised using the Marine Nature Conservation Review (MNCR) SACFOR scale of abundance (JNCC, 2017).

Sediment infauna

To determine the distribution and quantity of food available to mobile communities from sediment infauna surrounding the piers, three sediment samples were taken using a 2.75 litre Van-Veen grab deployed from a small boat at each sampling station (Figure 4.2). Macrofauna were sieved through a 0.5 mm mesh and preserved in 10% formalin in sea water. In the laboratory, macrofauna were counted and identified to species or lowest taxonomic level possible.

The organic content of a sub-sample from each sediment core was measured using the Difference-On-Ignition (DOI) procedure. A pre-weighed sample of dried sediment was placed in a muffle furnace at 450 ^{O}c for 8 hours and the weight loss translated in to the organic content of the sample (Luczak et al., 1997).

Stable isotope analysis

SIA of fish and their putative food items was applied here to determine if the species available on or around the pier were creating a preferred foraging area. A comparative study of blood and dorsal muscle samples was conducted using three fish species a) bass, *D. labrax,* a relatively large facultative piscivore; b) sole*, Solea solea,* a benthic scavenger; and c) pouting, *Trisopterus luscus,* a relatively small bodied predatory Gadoid. The rationale for using blood and tissue samples was to obtain an understanding of the temporal variation in the fish' diets. Muscle has a longer isotopic turnover rate, providing detection of long term dietary shifts (Pinnegar & Polunin, 1999; Post, 2002). In contrast, blood has a shorter isotopic turnover rate, which equates to a faster equilibrium being reached after a diet shift (Herzka, 2005; Ankjærø et al., 2012). Blood has shown to be accurate at detecting short term dietary changes in most animals, irrespective of their size (Bauchinger & McWilliams, 2009). Thus, the combination of blood and muscle samples should provide an increased understanding in the temporal dynamics of the consumer prey relationships of these fishes than one tissue alone (Phillips & Eldridge, 2006).

Samples of *D. labrax, S. solea* and *T. luscus* were collected by anglers during the evenings of the 9th July 2016 at Boscombe Pier and 6th August 2016 at Bournemouth Pier. Fish samples were collected during the summer months in order to obtain the species required and during the evening as they are nocturnal foragers. Ideally samples would be collected on the same date however this was unfeasible during the current study. All fish were weighed and measured to fork length, and euthanized via over-anaesthetic of Tricaine methanesulfonate (MS-222). Then, ~1ml of whole blood was collected using the heart puncture method by a needle and syringe and transferred to sterile Eppendorf tubes. Concurrently, white muscle from below the dorsal fin was also sampled and transferred to Eppendorf tubes (Ankjærø et al., 2012).

In order to quantify the putative food resources around the pier, samples of small fish (Sand eel- *Ammodytes tobianus,* Smelt- *Osmerus eperlanus)* and invertebrates *(Palaemon* sp*, Crangon crangon, Diogenes pugilator, Carcinus maenas, Crepidula fornicata)* were collected using a 30 m long beach seine net adjacent to both piers (~2 m water depth). Samples of blood and white muscle were sampled from the small fish and either tissue samples or whole specimens were used for the invertebrates (depending on body size). The remaining sources were sampled by scraping organisms from the pier pilings, *(Mytilus edulis, Nucella lapilus, Ulva* sp*, Porphyra* sp*.)* alongside 5 x 3 minute plankton trawls which were conducted from a boat around the piers. The polychaete and amphipod samples collected from the benthic sediment grab collect around the piers were also used as potential food sources. All samples of consumers and sources were dried in an oven at 60°C for 48 hours before transportation to the laboratory.

The samples were then submitted to the Cornell University Stable Isotope Laboratory, New York, USA, for analysis of $\delta^{13}C$ and $\delta^{15}N$. The tissues were ground to powder, with approximately 0.5 mg weighed out into a tin cup and the actual weight recorded using a Sartorius MC5 microbalance to ~1000 µg. The samples were then analysed on a Thermo Delta V isotope ratio mass spectrometer (Thermo Scientific, Waltham, MA, USA) interfaced to a NC2500 elemental analyser (CE Elantach Inc., Lakewood, NJ, USA). These were verified for accuracy against internationally known reference materials, whose values are determined by the International Atomic Energy Agency (IAEA; Vienna, Austria), and calibrated against the primary reference scales for $\delta^{13}C$ and $\delta^{15}N$ values. The accuracy and precision of the sample runs were tested after every 10 samples using a standard animal sample (mink) to compensate for possible machine drift and as a quality control measure; the overall standard deviation was 0.11 ‰ for $\delta^{15}N$ values and 0.09 ‰ for $\delta^{13}C$ values. Linearity correction was carried out to account for differences in peak amplitudes between sample and reference gases (N_2 or CO_2); the analytical precision associated with the $\delta^{15}N$ and $\delta^{13}C$ sample runs were estimated at 0.42 and 0.15 ‰, respectively. Final outputs were values of $\delta^{13}C$ (energy source indicator) and $\delta^{15}N$ (trophic level indicator), expressed as their isotope ratios per thousand (‰).

Stomach content analysis

Stomach content analyses were conducted on all fish specimens collected. The contents of the entire intestine were examined using a dissection microscope and the number of empty stomach was noted throughout. Prey items were identified to the lowest possible taxonomic resolution and grouped into nine categories according to the level at which they were identified: Amphipoda, Decapoda, Other Crustacea (all other crustaceans not included in the two preceding groups or could not be identified due to digestion), Bivalve, Polychaete, Fish, Plankton, Algae and Other (prey items not included in the 8 previous groups). The results of the stomach contents were used to inform the stable isotope mixing models.

4.2.3 Statistical Analyses

Baited Remote Underwater Video

The BRUV videos were viewed using VLC media player and split into one minute sections. A 5 minute settlement period was allowed before analysis commenced. For each deployment, species were identified to species level where possible. The metrics recorded from each deployment were *MaxD* (maximum number of species seen at any time during the 20 min deployment)*, MaxN* (maximum number of individuals of a species seen at any time during the 20min*)* (Cappo et al., 2004; Folpp et al., 2013). Previous studies have shown that both *MaxD* and *Max*N have been good indicators of relative fish abundance (Willis et al. 2000; Willis & Babcock, 2000). ANOVA was used to test for differences between site (Bournemouth, Boscombe), distances away from the pier (1m, 10m, 25m) and distances along the pier (Inner, Middle, Outer) in mean *MaxD* and *MaxN* using R (Version 3.3.2). Differences in species communities between site, distance away from the pier and distance along the pier were analysed using an ANOSIM and Multidimensional Scaling plot (MDS) on a squared root transformed Bray Curtis similarity matrix in PRIMER-E (Version 6) (PRIMER-E, 2001).

Sediment infauna samples

ANOVA was used to test differences between site, distance away and distance along the pier for species richness (S), total abundance (N) and species diversity (H) of the benthic fauna. ANOSIM was used to determine any variation in communities due to distance along or distance away from the pier followed by an MDS plot to illustrate the variation amongst samples. Finally the organic content of the sediment was analysed using a two-way ANOVA with ‘distance ‘away’ and ‘distance along’ as main factors.

Stable isotope analysis

Bayesian mixing models were performed using MixSIAR GUI in R (R Core team, Version 3.3.2) to determine proportions of potential prey items which contributed to the diet of consumers around the piers. To correct for isotopic fractionation between resources and consumers, species-specific and tissue-specific fractionation factors between the fishes and their prey were used where available (Table A2.1). All models were run for the normal run length (chain length: 100,000 iterations with burn-in of 50,000, with posterior thinning (thin: 50) and 3 chains). Model diagnostics were based on the Gelman-Rubin and Geweke, and converged sufficiently to accept the results (Stock & Semmens, 2013).

Potential prey items with similar taxonomic groups and isotope values were grouped to simplify the interpretation of results (Phillips et al., 2005). The source groups for secondary consumers*, D. labrax, T. luscus* and *S. solea,* were 'Molluscs' *(C. fornicata, M. edulis, N. lapilus),* 'Other Crustacea' *(Amphipoda, C. crangon, Palaemon sp.), Paguridae (D. pugilator),* 'Pisces' *(O. eperlanus, A. tobianus), 'Portunidae' (C. maenas)* and 'Polychaete' (*Nephtys cirrosa).* The source groups for the primary consumers*, O. eperlanus* and *A. tobianus* were 'Algae' *(Ulva sp., Porphyra sp.), '*Other Crustacea' *(Amphipoda, C. crangon, and Palaemon sp*.), 'Pisces' *(O. eperlanus, A. tobianus),* 'Plankton', and 'Polychaete' *(N. cirrosa).*

Both blood and tissue samples from consumers were analysed within the mixing models. As Bayesian models are very sensitive to discrimination factors (Bond & Diamond, 2011) the most appropriate discrimination factor was applied to each model (Table A2.1). For the fish primary consumers *(O. eperlanus* and *A. tobianus)* only blood samples were used in the mixing models due to financial constraints, using appropriate discrimination factors (Table A2.1).

Stomach contents analysis

In order to determine what percentage of the consumers diet consisted of specific food resources two equations were used. The vacuity index (VI, %) was calculated to determine how many stomachs were empty and the relative frequency of occurrence (rFO, %) were calculated to measure diet composition according to Hureau (1970) and Berg (1979)

a) $VI = Nv/Ne \times 100$ b) $rFO = Ndi/Nnv \times 100$

Where Nv= number of empty stomach, Ne= total number of examined stomach, Ndi= number of prey item x, Nnv= total number of prey items in all stomachs (adapted from Fehri-Bedoui et al., 2009).

4.3 Results

4.3.1 Baited Remote Underwater Video

Mobile species at different distances away from the pier

The BRUV data revealed that Bournemouth Pier had a higher overall mean abundance of species compared with Boscombe Pier. At Bournemouth Pier, there was a significant decrease in species richness with distance away from the pier, particularly between 1 m and 25 m (Figure 4.3, Table 4.2). Yet no significant differences in total abundance of species were recorded with varying distance away from the pier at either site (Table 4.2).

Table 4.2 2-way ANOVA for comparison of *MaxN* and *MaxD* with distance away from the pier (1 m, 10 m, 25 m) at a) Bournemouth and b) Boscombe Pier (***=P<0.001, **=P<0.01,*=0.05 NS= Not significant).

a) Bournemouth Pier

	MaxN			***MaxD***		
Source of Variation	df	F	p	df	F	p
Distance away	2	1.467	NS	2	4.883	*
Residuals	51			51		

b) Boscombe Pier

	MaxN			***MaxD***		
Source of Variation	df	F	p	df	F	p
Distance away	2	0.66	NS	2	0.285	NS
Residuals	48			42		

At Bournemouth Pier the assemblage composition was also significantly different between distances away from the pier (ANOSIM, Global R=0.203, P=0.001). Pairwise tests revealed significant differences between 1 and 10 m (P=0.001) and 1 and 25 m (P=0.001, Table 4.3). SIMPER revealed that 92% of the overall 39.7% similarities between samples at the 1 m stations at Bournemouth Pier were due to four species*; D. pugilator, Pomatoschistus minutes, D. labrax* and *C. maenas,* while the 10 m and 25 m stations were dominated by *D. pugilator* and *Nassarius reticulatus* (Table 4.3). There was no significant variation in communities with varied distance away from the pier at Boscombe. The MDS plots illustrate the variation in community similarity between distance away from the pier at Bournemouth and Boscombe Pier (Figure 4.4).

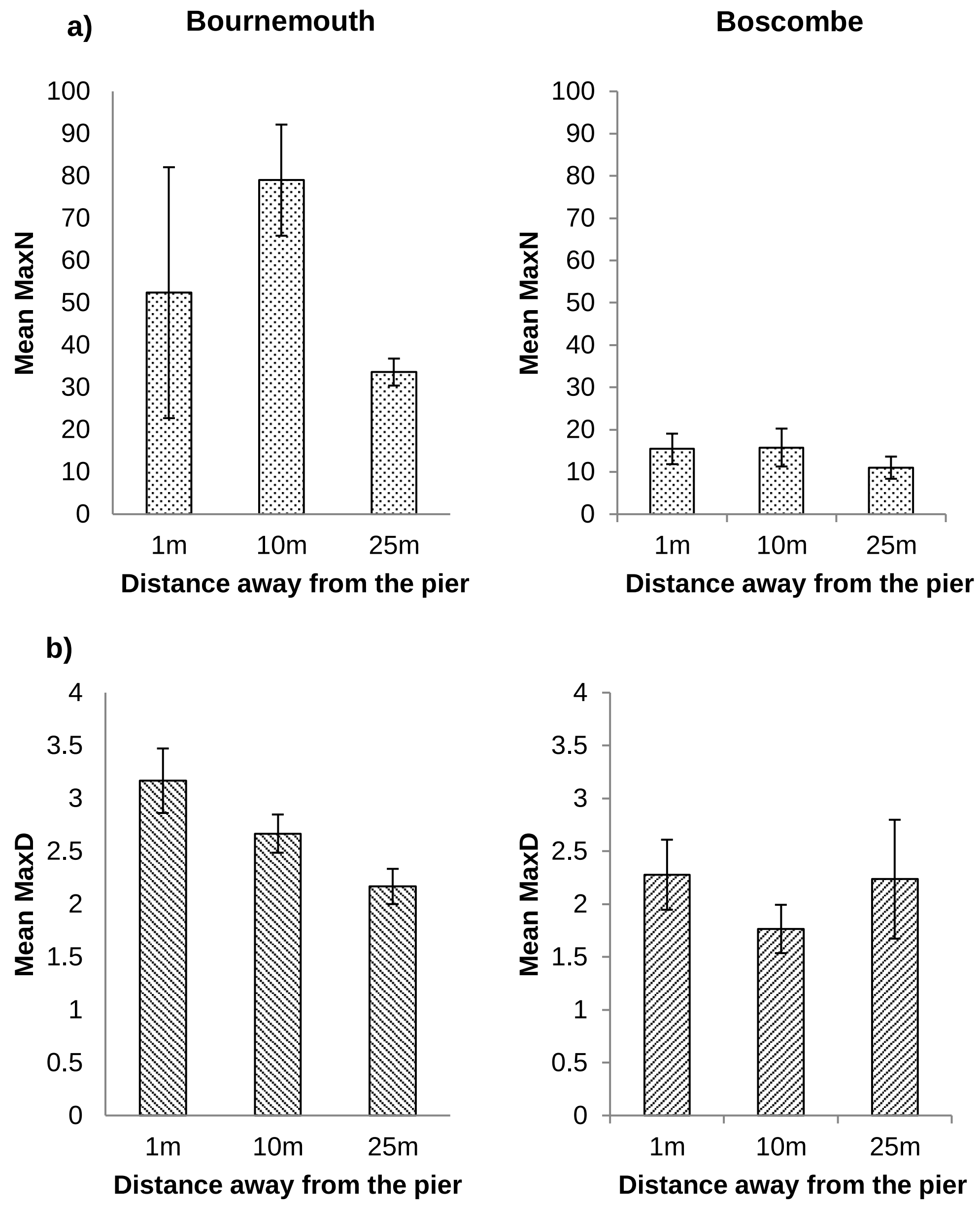
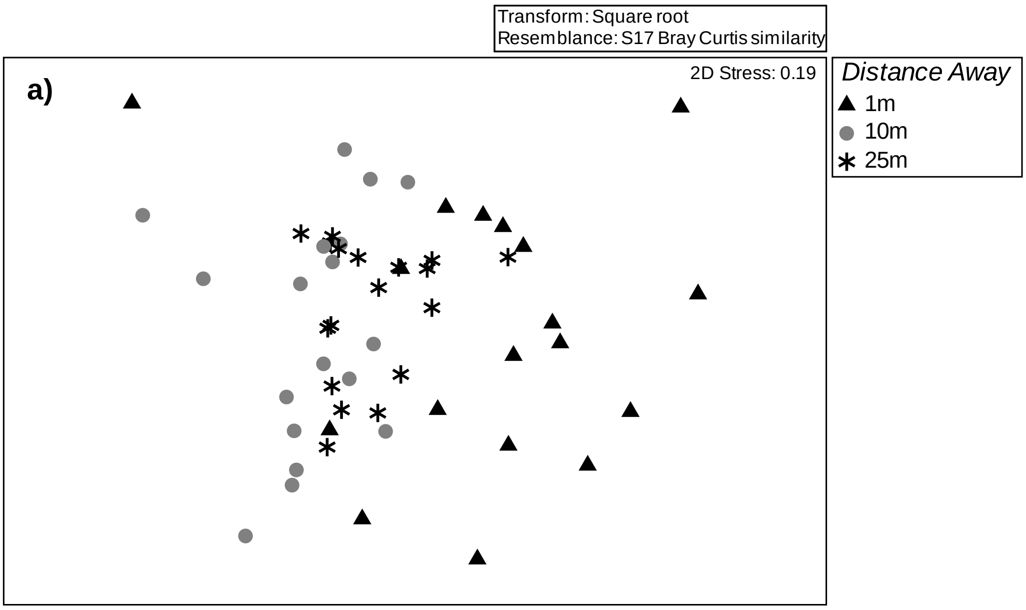


Figure 4.3 a) Mean *MaxN* (Maximum number of individuals) and b) Mean *MaxD* (Maximum number of species) with varied distance away from the pier, 1 m, 10 m, 25 m at Bournemouth and Boscombe Pier (Total N=18 for each pier, all distances along were combined) (+/- S.E.).



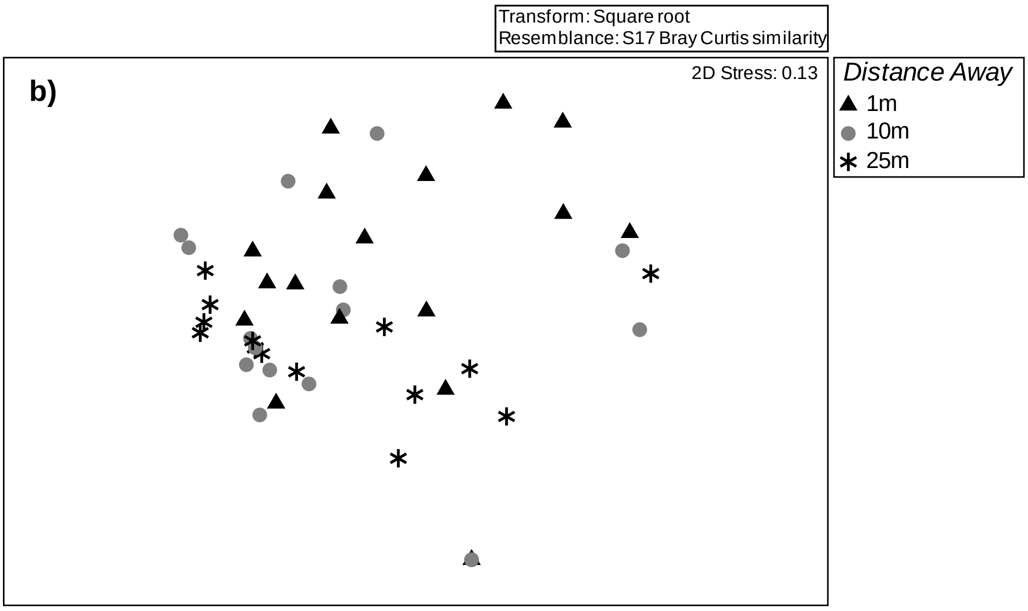


Figure 4.4 Multidimensional Scaling (MDS) plots of the communities recorded with varied distance away from the pier at a) Bournemouth Pier and b) Boscombe Pier during June-September 2015.

Table 4.3 Mean abundance of mobile fauna from BRUV deployments at 1 m, 10 m and 25 m away from a) Bournemouth Pier and b) Boscombe Pier (*MaxN* is the maximum number of individuals seen in any deployment).

a) Bournemouth Pier

		Mean *MaxN* (SD)		
Phylum	**Species**	**1m**	**10m**	**25m**
Chordata	*Raja* sp.	0.00	0.00	0.00
	Chelon labrosus	0.33 (0.97)	0.00	0.00
	Pomatoschistus minutus	1.33 (1.41)	0.33 (0.59)	0.38 (0.60)
	Osmerus eperlanus	3.61 (10.84)	6.61 (19.41)	0.00
	Dicentrarchus labrax	3.22 (6.74)	5.16 (13.30)	0.00
	Echiichthys vipera	0.05 (0.23)	0.05 (0.23)	0.11 (0.32)
	Ammodytes tobianus	28.00 (117)	0.00	0.00
	Spondyliosoma cantharus	0.00	0.00	0.00
	Trisopterus luscus	0.05 (0.23)	0.00	0.00
	Paralichthys dentatus	0.00	0.00	0.05 (0.23)
	Mullus surmuletus	0.00	0.00	0.00
	Gadus morhua	0.11 (0.32)	0.00	0.00
	Callionymus reticulatus	0.27 (0.57)	0.11 (0.32)	0.00
	Other small fish	1 (2.56)	11.11 (47.10)	0.05 (0.23)
Crustacea	*Diogenes pugilator*	14.27 (14.0)	37.0 (18.41)	28.77 (13.64)
	Carcinus maenas	0.44 (0.51)	0.22 (0.64)	0.33 (0.59)
	Liocarcinus marmoreus	0.05 (0.23)	0.16 (0.51)	0.05 (0.23)
	Necora puber	0.05 (0.23)	0.00	0.00
	Maja brachydactyla	0.5 (0.92)	0.05 (0.23)	0.05 (0.23)
Mollusca	*Tritia reticulata*	5.72 (16.35)	21.34 (34.54)	5.38 (9.38)
	Sepia officinalis	0.00	0.05 (0.23)	0.00
Cnidaria	*Rhizostoma pulmo*	0.05 (0.23)	0.11 (0.32)	0.05 (0.23)

b) Boscombe Pier

		Mean *MaxN* (SD)		
Phylum	***Species***	**1m**	**10m**	**25m**
Chordata	*Raja* sp.	0.00	0.00	0.06 (0.25)
	Chelon labrosus	0.00	0.00	0.00
	Pomatoschistus minutus	0.44 (1.14)	0.18 (0.40)	0.05(0.24)
	Osmerus eperlanus	1.66 (4.95)	0.00	0.00
	Dicentrarchus labrax	5.16 (11.54)	6.25 (18.57)	0.23 (0.75)
	Echiichthys vipera	0.00	0.12 (0.34)	0.00
	Ammodytes tobianus	0.00	0.00	0.00
	Spondyliosoma cantharus	0.27 (1.17)	0.00	0.00
	Trisopterus luscus	0.44 (1.88)	0.00	0.00
	Paralichthys dentatus	0.00	0.00	0.00
	Mullus surmuletus	0.16 (0.70)	0.00	0.00
	Gadus morhua	0.00	0.00	0.00
	Callionymus reticulatus	0.11 (0.32)	0.18 (0.40)	0.00
	Other small fish	0.22 (0.73)	0.06 (0.25)	0.00
Crustacea	*Diogenes pugilator*	9.16 (13.61)	10.87 (11.64)	10.05 (11.46)
	Carcinus maenas	0.22 (0.54)	0.12 (0.34)	0.05 (0.24)
	Liocarcinus marmoreus	0.22 (0.42)	0.18 (0.40)	0.47 (0.79)
	Necora puber	0.05 (0.23)	0.00	0.00
	Maja brachydactyla	2.83 (6.74)	0.56 (1.31)	0.35 (0.86)
Mollusca	*Tritia reticulata*	0.05 (0.23)	0.81 (2.28)	0.35 (0.78)
	Sepia officinalis	0.11 (0.33)	0.12 (0.34)	0.11 (0.33)
Cnidaria	*Rhizostoma pulmo*	0.05 (0.23)	0.00	0.00

Distance away from the pier- fish species only

In order to investigate the distribution for fish independently of other mobile fauna, separate analysis was conducted. For fish species alone, at Bournemouth Pier there was a significant difference in mean maximum species richness with distance away from the pier ($ANOVA_2$=11.23, $P<0.001$, Figure 4.5) and the pairwise test revealed a significant difference between the 1m and 10m stations ($P<0.01$) and the 1m and 25m stations ($P<0.001$). At Boscombe Pier there was also a significant difference in maximum species richness with distance away from the structure ($ANOVA_2$=5.95, $P=0.005$) and the pairwise test revealed differences between the 1m and 25m stations ($P<0.01$). Yet no significant difference in MaxN was revealed between varied distance away from the pier at either Bournemouth or Boscombe ($P>0.05$).

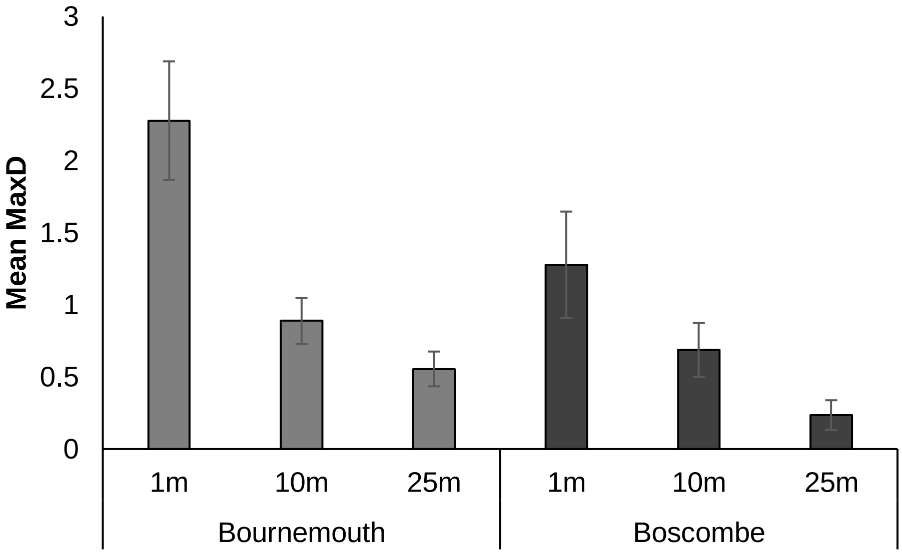


Figure 4.5 Fish mean maximum species richness (*MaxD*) for Bournemouth and Boscombe Piers with varied distance away from the pier (+/- S.E.).

Mobile species at different distances along the pier

Bournemouth pier showed no significant different in the mean maximum abundance or species richness with varied distance along the pier. At Boscombe Pier there was no significant differences in total abundance of species along the pier yet there was a significant difference in species richness (all species) with distance along the pier with the outer stations having significantly greater species richness than the middle and inner sections (Figure 4.6, Table 4.4).

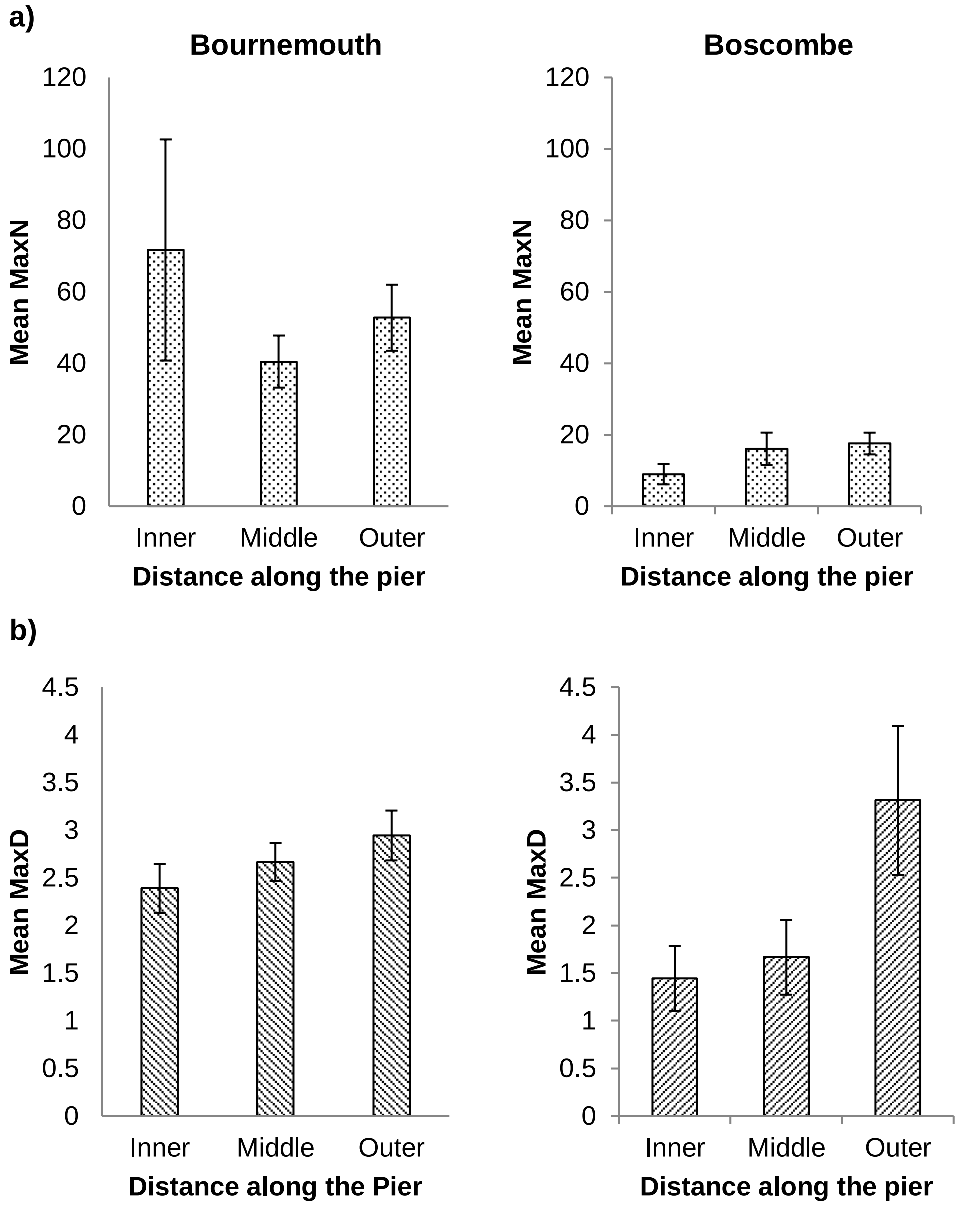


Figure 4.6 a) Mean *MaxN* (Maximum number of individuals in any video frame during deployment and b) Mean *MaxD* (Maximum number of species in any video frame during deployment) with varied distance along the pier: Inner, Middle and Outer at Bournemouth and Boscombe Piers (Total N=18 at each pier all distances away were combined) (+/- S.E.).

Table 4.4 One-way ANOVA for comparison of *MaxN* and *MaxD* with distance along the pier (Inner, Middle, Outer) at a) Bournemouth and b) Boscombe (*** = P<0.001, ** = P<0.01,* = 0.05, NS= Not significant).

a) Bournemouth Pier

	MaxN			***MaxD***		
Source of Variation	df	F	p	df	F	p
Distance along	2	0.678	NS	2	1.331	NS
Residuals	51			51		

b) Boscombe Pier

	MaxN			***MaxD***		
Source of Variation	df	F	p	df	F	p
Distance along	2	1.988	NS	2	10.45	***
Residuals	48			48		

There was no significant variation in communities with increased distance along the pier at Bournemouth. At Boscombe Pier, however, the communities did significantly vary between distances along the pier (ANOSIM, Global R=0.125, P=0.003), specifically at the outer and inner stations (P=0.002, Table 4.5). The SIMPER revealed that the Inner and Outer stations were 76% dissimilar; the inner stations were dominated by *D. pugilator* and *Liocarcinus marmoreus* while the outer stations were dominated by *D. pugilator, Maja brachydactyla, N. reticulatus, P. minutus* and *Rhizostoma pulmo* (Table 4.5). The MDS plots illustrate the variation in community similarity between distance along the pier at Bournemouth and Boscombe Piers (Figure 4.7).

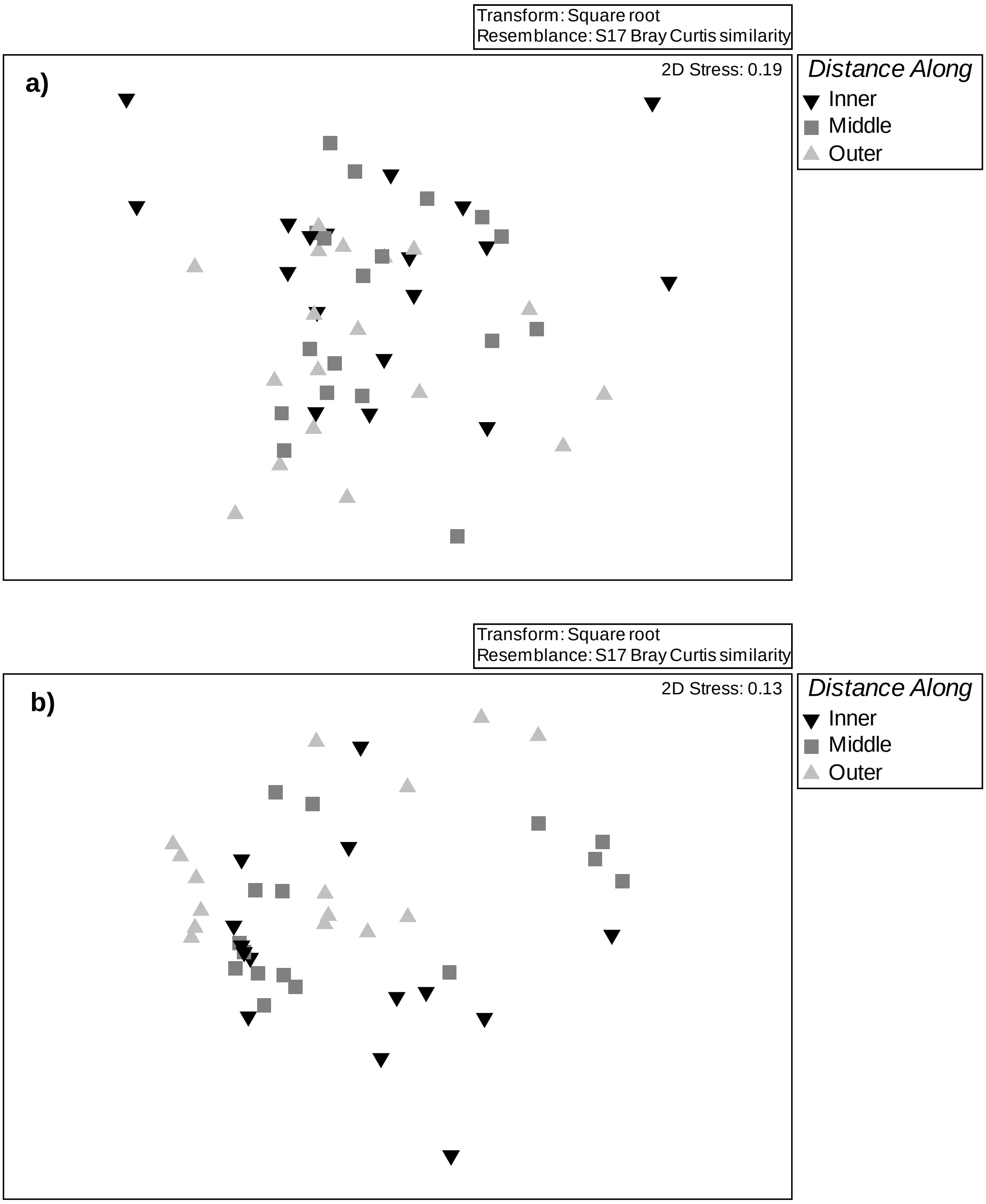


Figure 4.7 Multidimensional Scaling (MDS) plots of the communities recorded with varied distance along the pier at a) Bournemouth Pier and b) Boscombe Pier during June-September 2015.

Table 4.5 Mean abundance of mobile fauna from BRUV deployments at the Inner, Middle and Outer stations on the piers (*MaxN* is the maximum number of individuals seen in any deployment).

a) Bournemouth Pier

Phylum	Species	*Mean MaxN (SD)*		
		Inner	**Middle**	**Outer**
Chordata	*Raja* sp.	0.00	0.00	0.00
	Chelon labrosus	0.05 (0.23)	0.00	0.27 (0.95)
	Pomatoschistus minutus	0.38 (0.69)	0.66 (0.84)	1.00 (1.41)
	Osmerus eperlanus	0.44 (1.88)	0.00	9.77 (21.20)
	Dicentrarchus labrax	3.28 (7.66)	5.61 (16.19)	0.85 (2.69)
	Echiichthys vipera	0.11 (0.32)	0.05 (0.23)	0.05 (0.23)
	Ammodytes tobianus	28 (117.80)	0.00	0.00
	Spondyliosoma cantharus	0.00	0.00	0.00
	Trisopterus luscus	0.05(0.23)	0.00	0.00
	Paralichthys dentatus	0.00	0.00	0.05 (0.23)
	Mullus surmuletus	0.00	0.00	0.00
	Gadus morhua	0.05 (0.23)	0.05 (0.23)	0.00
	Callionymus reticulatus	0.05 (0.23)	0.11 (0.32)	0.22 (0.54)
	Other small fish	11.83 (47.01)	0.27 (1.17)	0.05 (0.23)
Crustacean	*Diogenes pugilator*	30.22 (22.42)	25.00 (15.77)	24.83 (15.18)
	Carcinus maenas	0.16 (0.38)	0.22 (0.42)	0.61 (0.77)
	Liocarcinus marmoreus	0.16 (0.51)	0.00	0.11 (0.32)
	Necora puber	0.00	0.05 (0.23)	0.00
	Maja brachydactyla	0.05 (0.23)	0.33 (0.84)	0.22 (0.54)
Mollusca	*Nassarius reticulatus*	2.33 (5.77)	11.77 (22.64)	18.72 (32.31)
	Sepia officinalis	0.00	0.00	0.05 (0.23)
Cnidaria	*Rhizostoma pulmo*	0.11 (0.32)	0.00	0.11 (0.32)

b) Boscombe Pier

Phylum	Species	**Mean MaxN (SD)**		
		Inner	**Middle**	**Outer**
Chordata	*Raja* sp.	0.00	0.05 (0.23)	0.00
	Chelon labrosus	0.00	0.00	0.00
	Pomatoschistus minutus	0.00	0.00	0.80 (1.21)
	Osmerus eperlanus	0.00	0.00	2.00 (5.39)
	Dicentrarchus labrax	2.50 (7.14)	7.00 (19.46)	1.73 (3.86)
	Echiichthys vipera	0.05 (0.23)	00.00	0.06 (0.25)
	Ammodytes tobianus	0.00	0.00	0.00
	Spondyliosoma cantharus	0.00	0.00	0.33 (1.29)
	Trisopterus luscus	0.00	0.00	0.53 (2.06)
	Paralichthys dentatus	0.00	0.00	0.00
	Mullus surmuletus	0.00	0.00	0.2 (0.77)
	Gadus morhua	0.00	0.00	0.00
	Callionymus reticulatus	0.00	0.00	0.33 (0.48)
	Other small fish	0.00	0.05 (0.23)	0.26 (0.79)
Crustacean	*Diogenes pugilator*	6.44 (11.52)	10.55 (9.71)	13.60 (14.70)
	Carcinus maenas	0.05 (0.23)	0.00	0.40 (0.63)
	Liocarcinus marmoreus	0.61 (0.77)	0.16 (0.38)	0.06 (0.25)
	Necora puber	0.00	0.00	0.06 (0.17)
	Maja brachydactyla	0.05 (0.23)	1.44 (2.59)	2.60 (7.10)
Mollusca	*Nassarius reticulatus*	0.00	0.00	1.33 (2.31)
	Sepia officinalis	0.00	0.00	0.14 (0.36)
Cnidaria	*Rhizostoma pulmo*	0.00	0.00	0.33 (1.29)

Distance along the pier- fish species only

Bournemouth pier showed no significant different in the mean maximum abundance or species richness for fish species with varied distance along the pier. At Boscombe Pier there was a significant difference in maximum species richness (*MaxD*) of fish species with distance along the pier ($ANOVA_2$= 10.67, P<0.001, Figure 4.8), the pairwise tests revealed significant difference between the Inner and Middle stations (P<0.001) and the Inner and Outer stations (P<0.001). There was also a significant interaction between distance away and distance along the pier at Boscombe ($ANOVA_4$= 3.07, P=0.02). Yet no significant difference in *MaxN* was revealed between varied distance along the pier at either Bournemouth or Boscombe (P>0.05).

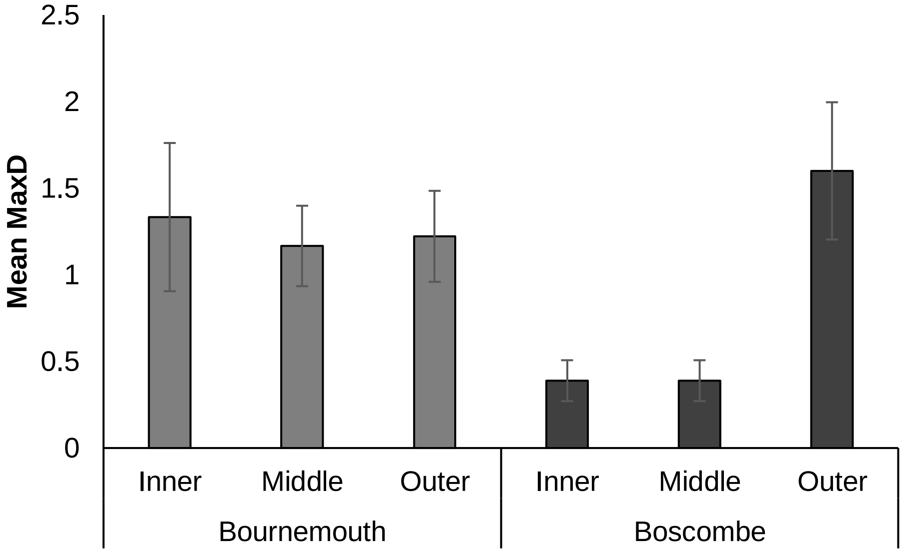


Figure 4.8 Fish mean maximum species richness (*MaxD*) for Bournemouth and Boscombe Piers with varied distance along the pier (+/- S.E.).

4.3.2 Epifaunal communities on pier piling

In total, nine main functional groups were recorded at both Bournemouth and Boscombe Pier, although the groups varied between sites (Table 4.6). Kelp was only recorded on the middle section of Bournemouth Pier and Ascidians were only recorded on the middle and outer sections of Boscombe Pier. The remaining groups were present on both piers. Barnacles, mussels and green algae covered the largest proportion of the pilings throughout with anemones only recorded on the middle and outer pilings and encrusting sponge only recorded on the inner and middle pilings.

Table 4.6 Abundance of epifauna on the pier pilings at Bournemouth and Boscombe Pier at the Inner, Middle and Outer sections according to the SACFOR scale (Superabundant (S)-80-100% coverage, Abundant (A)-40-79% coverage, Common (C)- 20-39% & coverage, Frequent (F) – 10-19% coverage, Occasional (O)- 5-9% coverage, Rare (R) 1-5% coverage).

	Bournemouth Pier			**Boscombe Pier**		
Functional group	**Inner**	**Middle**	**Outer**	**Inner**	**Middle**	**Outer**
Anemones (Cnidaria: Anthozoa)	-	O	F	-	F	O
Barnacles (Crustacea: Cirripedia)	S	A	S	S	S	S
Encrusting sponge (Porifera)	O	R	-	-	R	
Gastropods (Mollusca)	O	O	O	F	O	O
Green algae (Chlorophyceae)	A	F	A	A	A	A
Hydroids (Cnidaria: Hydrozoa)	C	F	O	O	F	F
Kelp (Ochrophyta)	-	R	-	-	-	-
Mussels (Mollusca: Bivalvia)	A	S	S	S	A	A
Red algae (Rhodophyceae)	C	O	A	O	C	C
Ascidians (Chordata: Tunicata: Ascidiacea)	-	-	-	-	R	R

4.3.3 Sediment infauna samples

The sediment infauna around the piers had significantly higher species richness (ANOVA$_1$ F=12.04, P<0.01), total abundance (ANOVA$_1$ F=45.07, P<0.001) and species diversity (ANOVA$_1$ F=6.07, P<0.05) at Bournemouth Pier compared to Boscombe Pier. Yet no significant difference was found with distance away from the pier or distance along the pier at either Bournemouth or Boscombe Pier (P>0.05 in all cases, Table

A2.2 Figure A2.4 & A2.5). In addition there was found to be no correlation between the abundance of sediment infauna and the abundance of mobile fauna along or away from the structure of either piers (Figure A2.6).

The assemblage composition was also significantly different between piers (ANOSIM, Global R=0.0.419, P=0.001). But there were no significant difference in the assemblages with increasing distance away from the pier and distance along the pier at either Bournemouth or Boscombe Pier. The MDS shows a clear separation in communities between Bournemouth and Boscombe Pier (Figure 4.9). The SIMPER revealed that 91% of the overall 66% dissimilarity between sites was due to nine species (Table 4.7).

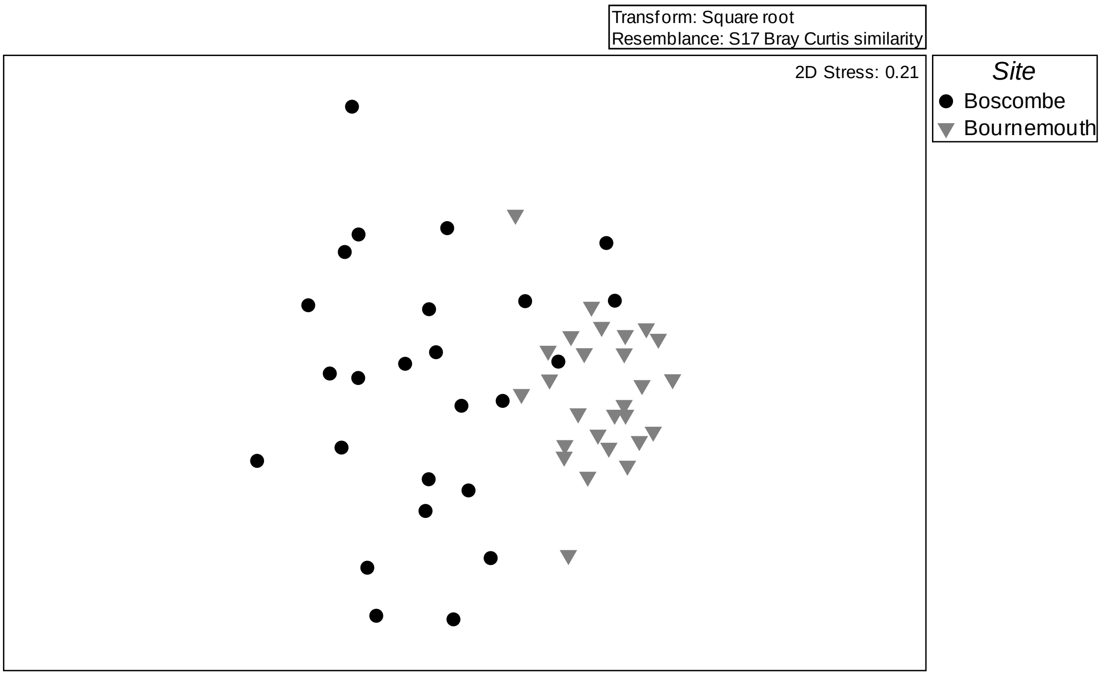


Figure 4.9 Multidimensional Scaling (MDS) plot of the benthic communities collected using a Van-Veen grab at each site (N=27).

Table 4.7 SIMPER analysis for benthic fauna collected from Bournemouth and Boscombe Pier: Bray-Curtis Dissimilarity between sites, Bold indicates highest average abundance (Bmth=Bournemouth, Bosc=Boscombe)

SIMPER: **Ave dissimilarity= 66.09**

Species/Group	Bmth Ave Abund.	Bosc Ave Abund.	Ave Dissimilarity	Contribution %	Cum %
Nephtys cirrosa	**4.13**	1.26	19.46	29.44	29.44
Bathyporeia sp.	**2.80**	1.33	12.12	18.34	47.78
Lanice conchilega	**1.33**	0.20	7.33	11.09	58.87
Phyllodoce maculata	0.80	**0.86**	6.45	9.76	68.63
Diogenes pugilator	**0.97**	0.25	6.05	9.16	77.79
Haustorius arenarius	0.25	**0.37**	3.08	4.67	82.46
Capitellidae	**0.33**	0.08	2.08	3.15	85.61
Mysida	0.18	**0.26**	1.93	2.92	88.53
Urothoe brevicornis	0.07	**0.20**	1.67	2.53	91.07

There was no significant difference in percentage organic content between both distance away from the pier ($ANOVA_{1,25}$ F=0.01, P>0.05) and distance along the pier ($ANOVA_{1,25}$ F=3.44, P>0.05).

4.3.4 Estimates of fish diet

Stomach content analysis

The stomach contents of the fish sampled collected were analysed to determine the main prey categories consumed from around the piers. The stomach contents of the *D. labrax* contained five food categories, with a relatively large proportion of other Crustacean (21%), other (15%) and Decapoda (15%) (Table 4.8, Figure A2.7). The stomach contents from the *T. luscus* contained six food categories, dominated by other Crustacea (40%) and Decapoda (19%). *S. solea* stomach contents contained three food categories of which Crustacea (29%) had the greatest relative frequency, whilst the stomachs of the *O. eperlanus* contained six food categories with a large proportion of other Crustacea (40%) being recorded. In contrast, only Decapoda (64%) were recorded in the stomachs of *A. tobianus* (Table 4.8, Figure A2.7). The results from the stomach content analysis were used to inform the Bayesian mixing models for the SIA and to determine the potential for epibiota on the pier pilings to be a viable source of food for mobile species.

Table 4.8 Relative Frequency of each prey category in the stomach contents of *D. labrux, T. luscus, S. solea, O. eperlanus* and *A. tobianus* (N=number of samples, VI = vacuity index).

	Species				
	D. labrax	*T. luscus*	*S. solea*	*O. eperlanus*	*A. tobianus*
Prey Category	N=10 VI=30%	N=7 VI= 0%	N=8 VI= 62%	N= 12 VI= 33%	N=11 VI= 36%
Algae	0.00	0.00	0.00	0.01	0.00
Mollusc (Bivalve)	0.00	0.04	0.01	0.00	0.00
Other Crustacea	0.21	0.40	0.29	0.40	0.00
Decapoda	0.15	0.19	0.00	0.00	0.64
Pisces	0.10	0.03	0.00	0.05	0.00
Plankton	0.00	0.00	0.00	0.01	0.00
Polychaete	0.10	0.09	0.08	0.01	0.00
Other	0.15	0.02	0.00	0.04	0.00

Stable isotope results

In general, the δ^{13}C and δ^{15}N results for both blood and muscle samples revealed step-wise enrichment through a series of trophic levels (Figure 4.10). These included the position of the primary consumer fishes, such as *O. eperlanus*, up to the secondary consumers *D. labrax* (Figure 4.10), suggesting that these *D. labrax* were predominantly piscivorous (Pickett & Pawson, 1994; Spitz et al., 2013).

The δ^{13}C and δ^{15}N values for the blood samples taken from the three fish species were consistently lower than the muscle samples, with the exception of δ^{13}C for *S. solea* (Figure 4.10). Both δ^{15}N and δ^{13}C revealed a wide range of variation for both blood and muscle samples in all secondary consumers (Figure 4.10). The primary consumers had lower δ^{13}C and δ^{15}N values then the secondary consumers as expected (Figure 4.10). Algae, being a primary producer, had the lowest δ^{15}N values, yet plankton had a much higher δ^{15}N values, perhaps the result of trophic enrichment of some predatory zooplankton species.

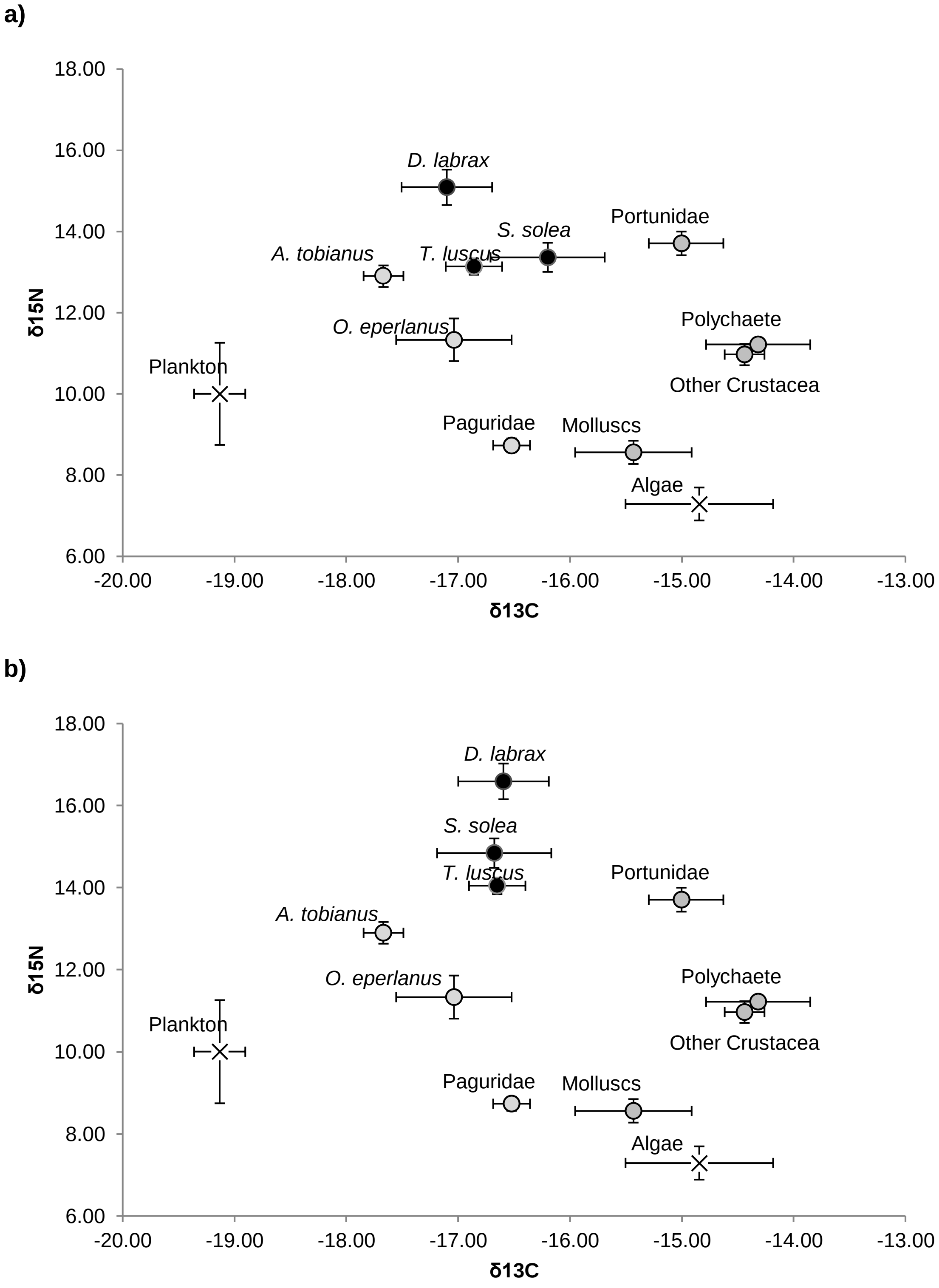


Figure 4.10 Mean δ^{13}C versus δ^{15}N of a) blood and b) muscle from secondary consumers (black circle), primary consumers (grey circles) and primary producers (black cross) (+/- SE).

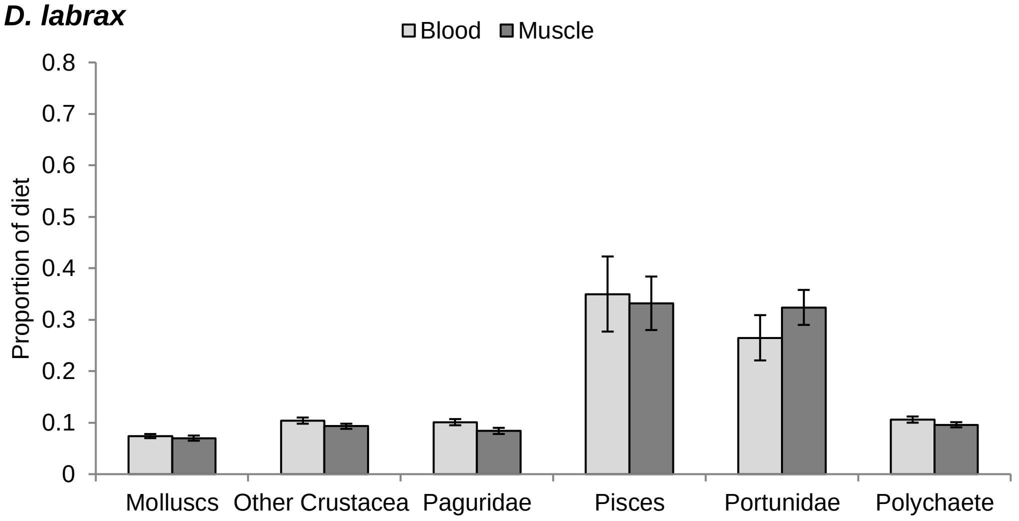
Bayesian Mixing Model results

The Bayesian mixing models predicted that the diet of *D. labrax* included all six prey categories, but with the two main prey sources being Pisces and Portunidae which are both found in higher abundance in close proximity to the piers (Figure 4.11a). The results from the stomach contents found five of the six prey categories, although no Molluscs were found. Their dietary proportions differed between the mixing models and the stomach contents, with a higher proportion of other Crustacea being recorded in the stomach contents (Figure 4.11, Table 4.8). The Bayesian mixing models for *T. luscus* corresponded largely to the results of the stomach contents analysis. However, the mixing models revealed Paguridae to have the largest proportion, whereas the stomach contents revealed a higher frequency of occurrence for other Crustacea (Figure 4.11, Table 4.8); both Paguridea and other Crustacea were found in greater abundances around the piers.

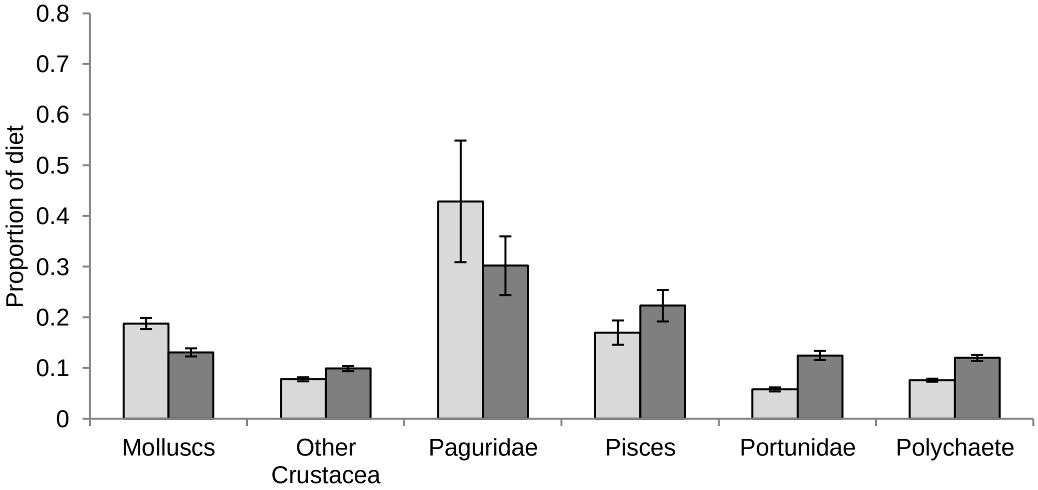
For *S. solea*, three prey categories were found in both the mixing models and the stomach contents; Molluscs, Polychaete and other Crustacea. The muscle samples revealed higher proportions of Polychaete and other Crustacea whereas the blood samples which represent the short-term diet, revealed a higher proportion of Molluscs, although the estimates had large variation (Figure 4.11, Table 4.8). Molluscs, particularly the bivalve *Mytilus edulis,* were found to be abundant on the pier pilings (Table 4.6) and absent on the surrounding seabed.

The mixing models for *O. eperlanus* revealed five prey categories with a high proportion of Algae and Plankton, whereas the stomach contents revealed three main prey categories with a large proportion of other Crustacea (Figure 4.12, Table 4.8). The mixing models for *A. tobianus* revealed five prey categories, with plankton and algae having the greatest contribution to diet, whereas the stomach contents found evidence of only one prey type, other Crustacea (Figure 4.12, Table 4.8). With the exception of Polychaete and other Crustacea all the remaining prey categories were observed either on the pier pilings or were found in greater abundance in close proximity to the pier (1m). The stomach content analysis and the stable isotope analysis highlighted the importance of small fish and crabs as prey resources for both the primary and secondary consumers.

a) ***D. labrax***



b) ***T. luscus***



c) ***S. solea***

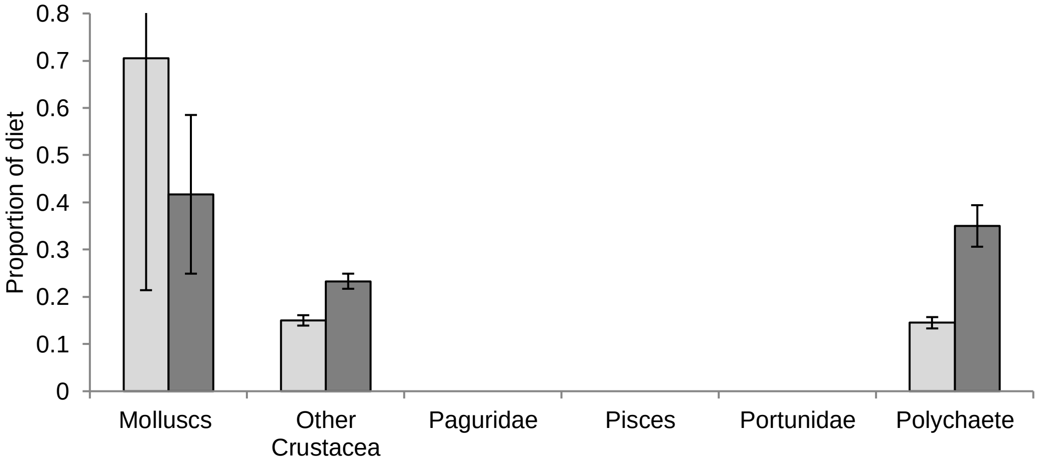


Figure 4.11 Comparison of blood and muscle samples to illustrate dietary proportions of secondary consumers a) *D. labrux* b) *T. luscus* and c) *S. solea* based on Bayesian mixing model (MixSIAR), Error bars are standard errors.

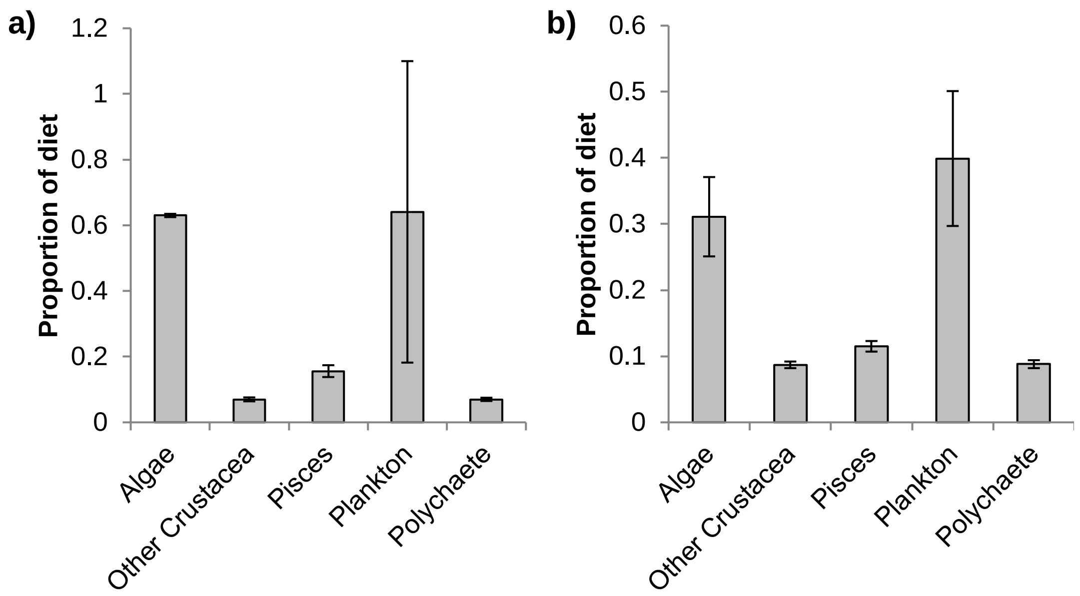


Figure 4.12 Dietary proportions of a) *O. eperlanus* and b) *A. tobianus* based on Bayesian mixing model (MixSIAR) from blood samples, error bars are standard errors (note differences in scale).

4.4 Discussion

The presence of submerged artificial structures such as piers has attracted mobile communities which would otherwise be absent. In Poole Bay, the provision of new hard substrate has enabled epibiota to colonise the pier pilings which has subsequently attracted fish and other mobile species to these local areas. The pier may act as a corridor, attracting species from deeper waters to the shallow intertidal zone, where adequate food resources can be acquired. Using stable isotope and stomach content analysis on the mobile species collected, strong links were discovered between consumers and prey within the food web around the piers. Understanding the mobile communities associated with artificial structures is important in order to appreciate how future installations of structures may influence the overall community dynamics of the region.

Proximity to the pier

An increased number of species were observed at the stations closest to the pier (Figure 4.1, Model A), which were predominantly fish species. This could be attributed to the protection provided by the structure from predation and adverse environmental conditions such as strong currents. As all sections of the piers were heavily colonised with epifauna during this study, it is hard to separate the effects of structural presence from the presence of epifauna. However, without the pier pilings there would be no hard substrate in which epifauna could colonise, suggesting that the epifauna and fauna may provide a secondary benefit to the pier pilings alone. Studies have shown that epifauna on pier pilings weakly correlates to the abundance of fish Coleman & Connell, (2006) and Hixon & Beets (1989) have shown that fish respond more to the presence of pier pilings than to the presence or absence of epifauna on the pilings. Throughout this study, fish were observed to be positioning themselves behind the pier pilings, which could be a foraging strategy or a means of refugia. Pelagic fish, such as bass, are known to use structures like piers to improve their feeding strategy, as prey species are often swept past in the currents created by the structures (Pickett & Pawson, 1994). This study found higher numbers of both predatory species, such as bass, and prey species, such as sand eel and smelt, present at the stations closest to the pier supporting this literature. Compared to the areas at 10 m and 25 m away from the piers, the piers supported a greater number of fish, suggesting that the pier pilings create an area of attraction. In addition, the overhead structure of the pier may have also created a de facto Marine Protected Area, an area which is off limits to human activity due to inaccessibility (Schroeder & Love, 2002).

Along the pier

At Boscombe Pier there was a greater number of all species at the outer end of the pier, confirming the hypothesis that species richness of mobile fauna would increase with distance along the pier (Figure 4.1, Model B). However this could be attributed to either the increased surface area of pier pilings at the outer end of the pier and/or increased water depth; Boscombe Pier is approximately 90 m shorter than Bournemouth and has a more pronounced variation in water depth between stations. The inner stations at Boscombe were extremely shallow (<2 m) compared with the middle and outer stations (2-4 m), whereas the stations at Bournemouth Pier were more consistent throughout the length of the structure (2-4 m).

The results from this study strongly support the hypothesis that species richness increases with increased water depth (Figure 4.1, Model B). An additional factor affecting the distribution of mobile fauna around artificial structures is the submerged surface area of the structure, as this could vary the quantity of epifauna and flora and potential shelter provided. Apart from greater water depth, the higher species richness at the outer end of Boscombe Pier could also be related to the increased number of pilings at the outer end resulting in a larger submerged surface area at the head of the pier. This was not the case for Bournemouth Pier as the pilings were of similar surface area throughout the length of the surveyed part of the structure. Species have shown to select habitats based on habitat complexity (Schofield, 2003; White et al., 2014; Lavender et al., 2017), which can be linked to surface area (Loke & Todd, 2016). The higher habitat complexity and submerged surface area observed at Bournemouth Pier could explain why it has a higher total abundance of mobile and benthic fauna when compared to Boscombe Pier. Another possible factor that may result in a larger number of species at the outer end of the pier is the reduction in anthropogenic disturbance (e.g. swimmers and surfers) from the shoreline (Schultz et al., 2014), however there are still high levels of angling which occur at the outer sections of both piers.

The use of BRUV to survey the communities around artificial structures has rarely been used; however in this study the technique has revealed high resolution in patterns of mobile species distribution. The effect of the bait plume has been debated in previous studies (Harvey et al., 2007; Dorman et al., 2012; Hardinge et al., 2013; Wraith et al., 2013) and has the potential to overestimate the number of species at any single point due to the lure of the bait, yet very few fish species were observed interacting with or feeding on the bait. The main bait consumers were benthic species, including the five species of crab (*D. pugilator, C. maenas, L. marmoreus, N. puber, M. brachydactyla*) and the netted dog whelk (*Tritia reticulata*). Furthermore the fact that patterns in *MaxD* and *MaxN* were observed at the different stations indicates that the bait plume did not attract additional species/individuals from the surrounding areas and that BRUV can characterise the distribution of mobile fauna at local scales. Comparative studies with un-baited cameras would be recommended in the future in order to determine the influence of the bait. It has been acknowledged that cryptic species, herbivorous species or night foragers may have been missed due to the nature and timings of sampling method (Clynick et al., 2007). In addition, due to the protocol of the BRUV sampling the BRUV system could only be used in adequate water visibility (>1 m), reducing the overall number of replicates obtained throughout the study.

Use of the piers by mobile fauna

Evidence suggests that the shelter provided by the piers can create a refuge from both environmental conditions such as currents and wave action and protection from predators, which is similar to the function of some artificial reefs (Rilov & Benayahu, 1998). Caddy (2007) found the prime attraction of fish to artificial reefs was the shelter provided from predators, which varies depending on habitat complexity. The shelter provided by the pier has been found to be relative to the size of the fish (Hixon & Beets, 1989; Coleman & Connell, 2006), therefore the surface area of the pier may influence the community composition of mobile fauna (Luckhurst & Luckhurst, 1978; McCormick, 1994; Metzger et al., 2001). This may explain why the larger Bournemouth Pier had a greater population of mobile fauna than Boscombe Pier.

The pilings in the current study were densely colonised by a variety of food sources including algae, mussels, gastropods and crustaceans, the majority of which were absent in the surrounding soft sediment habitats due to a lack of hard substrata.

Molluscs, were both abundant on the pier pilings and recorded in the stomach contents of *S. solea,* which supports current literature that suggests *M. edulis* is a common prey source (Pickett & Pawson, 1994; Reubens et al., 2011; Spitz et al., 2013). As *M. edulis* is not present on the sandy habitats surrounding the pier, this suggests that the pier could be providing a prey resource which would typically be absent without the pier pilings. Subtidal *M. edulis* populations in natural habitats may exist elsewhere within Poole Bay and beyond, however due to time and financial constraints no surveys were carried out or samples taken from other areas. Thus, no comparison could be conducted to determine if consumers were solely eating around the pier.

The stomach content analysis and the stable isotope analysis highlighted the importance of small fish and crabs as prey resources for both the primary and secondary consumers. The BRUV surveys found increased numbers of small fish species in close proximity to the pier pilings, suggesting that small pelagic fish such as *O. eperlanus* and *A. tobianus* may be attracted to the epibiota on the pier pilings as either a refuge or food resource (Wickham et al., 1973). This in turn could attract predators, such as *D. labrax*, to the pier due to the increased food availability of small fish. Yet a substantial amount of the diet of *O. eperlanus* consisted of zooplankton, suggesting they might also be utilising the more turbulent conditions created around pier pilings to feed on the plankton in the currents (Pickett & Pawson, 1994; Rilov & Benayahu, 1998). Although the distribution of fish was influenced by the pier structure, the smaller crustaceans found in the stomach contents, such as amphipods and polychaetes, were commonly found in the soft sediment habitats. Therefore the presence of these fish species in the vicinity of the structure is unlikely to be attributed to the presence of food. Studies have suggested that fish forage in the areas adjacent to artificial structures (Randall, 1963; Lindquist & Cahoon, 1994; Coleman & Connell, 2006), avoiding the areas underneath piers due to the negative effect of shading (Able et al., 2013; Munsch et al., 2014). Yet in the current study, there was an even distribution of food available in the sediment around the pier (Figure 4.1, Model C), suggesting that the fish had the potential to forage both on the pier and in the areas surrounding the pier. Some studies have shown that the seabed underneath the pier can be organically enriched due to the debris of shells and organic matter falling from the pier pilings (Wilhelmsson et al., 2006) which can attract scavengers, such as crabs and flatfishes. This attraction of additional foragers would increase the species richness in close proximity to the pier. However, whilst several species of crab were recorded on and around the pier pilings, there was no significant increase in fish scavengers detected.

Although SIA is a useful tool in determining the trophic position of species collected from around the pier, the interpretation of a consumer's diet through mixing models is highly dependent on accurate knowledge of fractionation rates for specific species and tissues- which vary both spatially and temporally (Phillips et al., 2014). For the majority of species the specific fractionation rate do not exist, therefore values from comparable species are used (Vander Zanden et al., 2015), which can reduce the accuracy of results. In addition Bayesian mixing models do not consider the availably of prey resources, instead they assume that all consumers eat the same proportion of prey which is unrealistic, however new models are being developed to incorporate this information (Yeakel et al., 2011). While SIA has shown to be reliable in closed and semi closed systems such as rivers and lakes (Post, 2002), open systems such as coastal areas can present problems, particularly in terms of collecting all potential sources for a consumer. SI mixing models are inaccurate when sources are missing or unknown (Phillips et al., 2014). As *D. labrux* is a transient species which forages in both deep and shallow waters it is likely that not all of the sources were sampled during this study. Future studies should aim to collect putative sources from control sites in order to compare the variation in source isotopes. Although Bayesian mixing models can account for uncertainty it is recognised that depending on the research question, it may not always be the best method to use.

Conclusions

This research has demonstrated the application of BRUV surveys and SIA to examine the mobile communities associated with piers and their potential influence in the food web. This study highlights the abundance and diversity of mobile fauna found in close proximity to the piers, which would otherwise be absent from soft sediment areas without these structures, the findings of which are applicable to windfarms, oilrigs and other submerged pilings. A greater number of mobile species particularly fish were observed at the stations closest to the structure and at the outer end of the pier. The benthic infauna distribution however was not seen to be influenced by the presence of the pier structure. The findings supported the idea that epifauna and smaller prey species found around the pier structures were being utilised as a food resource, therefore contributing to the food web.

This research indicates that structures built with a high level of structural complexity and which extend to at least 6m water depth will attract the greatest diversity of species. Future work should focus on separating the effect of structural complexity and substrate composition in order to determine which factor is most influential in attracting species to the area. Seasonal variation should also be considered especially to record nocturnal or migratory species which may have been overlooked in the current study. Experiments should also be conducted into the effect of the turbulent flows created around the pier pilings in providing an enhanced foraging area. In light of the continued addition of artificial structures within the coastal zone, understanding the effect structures have on the distribution of mobile species and provision of food resources is vital in understanding the connectivity of marine communities and the consequences they might have on multiple trophic levels.

5 Do Inshore Structures act as Stepping Stones for Dispersing Native and Non-native Species?

Abstract

The connectivity of fragmented habitats is important to understand in order to conserve and manage marine ecosystems. Hydrodynamic and particle tracking models were used to assess the potential connectivity between natural shores and artificial structures within Poole Bay, Dorset. The model predicted that artificial structures do have the potential to become stepping stones for the dispersal of both native and non-native species within Poole Bay, which was partially supported by field observations of distribution. Pelagic larval duration was seen to have an effect on the dispersal ability, with species having longer pelagic larval durations being transported furthest. Due to the complex hydrodynamic nature of Poole Bay there was very little model larval retention and a high proportion of particles were lost within the first 24 hours from release, which could explain poor settlement of some climate migrant species in the region. Yet importantly, larvae of non-native species introduced into Poole Harbour with short pelagic life history stages, for example the tunicate *Styela clava,* have the potential to reach natural rocky shores within 24 hours, which could result in settlement. The models created for this chapter will help identify key habitats and potential stepping stones which can help improve the understanding for future coastal management and marine spatial planning.

5.1 Introduction

Understanding the connectivity between artificial structures and natural habitats is crucial in order to appreciate the ecological impact they have on marine communities. The construction of artificial structures in areas which lack natural hard substrate have the potential to alter the distribution and extend the natural dispersal ability for certain marine species including non-native and climate migrants as mentioned in Chapter 1 and 2 (Bulleri et al., 2006; Vaselli et al., 2008; Keith et al., 2011; Adams et al., 2014; Airoldi et al., 2015). The majority of marine species produce pelagic propagules, either larvae or spores, which are transported in the water column by tidal and wind-driven currents. The distance larvae can travel is linked to the time spent in the water column (Shanks et al., 2003), which is known as the pelagic larval duration (PLD) and this varies between species and environmental conditions. Understanding the hydrodynamic forces behind larval dispersal is key in order to interpret species distribution and the potential connectivity between populations.

As ports and harbours are known points for the introduction of non-native species it is important to understand how introduced species could be transported around the region. The connectivity between introduction points and natural shores is important to recognise as wide spread effects could occur. In addition, the location of artificial structures within the region is also important to understand as these too could influence the dispersal of marine organisms. Artificial structures can create barriers to larval dispersal; or alternatively can facilitate dispersal by acting as stepping stones to seed areas further along the coast (Bulleri & Airoldi, 2005; Adams et al., 2014). Stepping stone introduction is the spread of a species from an original site to new areas through a series of intermediate areas and or structures (stepping stones) (Clarke & Therriault, 2007). If two sites were previously isolated then a new connection could have an influence on the population size and structure at both sites. Stepping stone movement occurs over multiple generations, an individual must settle and reproduce in order for its offspring to be transported to a new area in which is can settle and subsequently reproduce itself and in turn expand the natural dispersal distribution. Artificial structures could become stepping stones between both natural and artificial habitats, potentially having an important impact on the metapopulation connectivity of a region. Species with short pelagic larval duration are thought to be more successful at settling in areas with stepping stones due to their shorter dispersal ability and increased habitat availability (Adams et al., 2014).

Larval dispersal

Marine larvae can have various characteristics and abilities, some larvae, for example of crustaceans and fish, have swimming ability which can change their vertical positioning behaviour in the water column (Sponaugle et al., 2002), this can impact their exposure to currents, especially near the bed and shoreline. Yet, due to their physical size, even larvae with swimming abilities are still greatly driven by hydrodynamic forces such as waves, currents and upwellings (Cowen et al., 2006; Paris et al., 2007). Physical features of the coastline and local oceanographic conditions are highly influential for the dispersal of larvae. Headlands are known to alter the water flow rates and retention levels in coastal areas, which may lead to an increased retention of larvae in bays (Graham & Largier, 1997; Lipphardt et al., 2006; Mace & Morgan, 2006). The south coast of England is a very complex coastline with various headlands and bays resulting in complex hydrodynamic conditions. Artificial structures situated along the coast may also alter the hydrodynamic currents along the shore and could subsequently alter larval dispersal patterns. The implications of constructing artificial structures on marine biodiversity is important to understand in order to inform future marine planning and sustainable development.

Hydrodynamic modelling

Hydrodynamic modelling can be used to simulate the hydrodynamic conditions within a modelled area, which can allow us to understand how far larvae could be transported. Hydrodynamic modelling coupled with particle tracking models uses the natural water processes such as currents, tides and wave energy to determine the potential and probability for dispersal of particles. These particles can represent any object influenced by physical processes for example planktonic larvae or sediment. Hydrodynamic modelling has been used in a variety of studies from modelling swimming aquatic animals (Willis, 2011) to modelling the dispersal of planktonic larvae (Pfeiffer-Herbert et al., 2007; North et al., 2009; Herbert et al., 2012; Narváez & Klinck, 2012; Adams et al., 2014; Bray et al., 2017).

Hydrodynamic models have been used in the context of marine resource management (James et al., 2002; Mitarai et al., 2008) and in particular looking at the connectivity of fish populations and the design of Marine Protected Area (MPA) networks (Jones et al., 2009). Few studies have modelled the climatic change impacts on larvae dispersal, although Keith et al. (2011) used hydrodynamic modelling to identify physical barriers of larvae dispersal for climate migrants, comparing the results to observed range expansions in the field. More recently, studies have focused on the connectivity between populations of non-native species on natural and artificial substrate, including the potential for artificial structures to act as stepping stones to facilitate their dispersal (Vaselli et al., 2008; Adams et al., 2014; Airoldi et al., 2015). Herbert et al. (2012) modelled the natural spread of non-native Manilla Clam from its introduction point within Poole Harbour and found that through hydrodynamic processes the larvae can be transported to the next available habitat outside the harbour by natural means. It is particularly important to understand habitat connectivity around ports and harbours which are known to be the most intensely human impacted areas of our coastline as well as the main introduction points for non-native species (Bax et al., 2003; Mineur et al., 2012).

Due to the increase in marine renewable energy devices being constructed in the marine environment there is the potential for these structures to become stepping stones for species dispersal. Adams et al. (2014) modelled that the structures could provide a habitat for pelagic larvae which would otherwise be lost offshore and which in turn creates a source population with the ability to reseed the natural coastal sites. The authors also noted that species with short pelagic larval durations will be affected greatest by the addition of new structures to the marine environment. Bray et al (2017) also used bio-physical models to assess the connectivity of areas within the Adriatic Sea for potential wind farm developments and were able to identify areas of high connectivity and areas of isolation which will be used to inform the construction of offshore structures.

The aim of this chapter is to determine the potential connectivity between natural and artificial habitats within Poole Bay. Do artificial structures provide stepping stones for marine larvae and other propagules to recruit and establish populations to extend the natural dispersal ability of their offspring or do they prevent dispersal by creating barriers?

The main hypotheses are:

(1) Larvae leaving Poole Harbour will not reach natural sites through modelled hydrodynamic processes;

(2) Larvae released from the two natural sites (Peveril Point & Swanage) will reach the artificial sites within Poole Bay;

(3) Pelagic larval duration will affect the distance travelled.

5.2 Methods

5.2.1 Study Sites

The study was conducted in Poole Bay on the south coast of England, between Swanage and Christchurch. The coastline has strong west and south-westerly prevailing winds, a double high water and a spring tidal range of 2 m (Humphreys, 2005). There are two harbours located within the study area; Poole Harbour to the west and Christchurch Harbour to the east. Intertidal habitats within the harbours are predominantly fine sands and clays whereas the dominant subtidal habitat within Poole

Bay is fine mobile sand and occasional rocky reef habitat (Herbert et al., 2012). The entrances to both Poole and Christchurch Harbours are very narrow creating fast flowing tidal streams (Humphreys, 2005). Poole and Christchurch Harbours contain various intertidal and subtidal artificial structures, including sheet piling, jetties, seawalls and boat moorings. Poole Bay has a long sandy beach (16 km) which has 80+ groynes constructed along its length in order to prevent coastal erosion. The groynes are variously constructed of wood and rock as mentioned in Chapter 2 and 3 and they provide the only hard substrate within the intertidal zone. Boscombe Artificial Surf Reef is also situated in the centre of Poole Bay adjacent to the coastline (Herbert et al., 2017) which predominantly provides subtidal artificial habitat.

5.2.2 Model Species

Four species of different larval durations were chosen to be modelled and observed; all species are found on hard substrata on both natural and artificial substrates.

The model species were:

1) the non-native sea squirt *Styela clava* with a larval duration of 24 hours (Clarke & Therriault, 2007). This species may be found both intertidally and subtidally.
2) the native intertidal gastropod *Gibbula umbilicalis* with a larval duration of 6 days (Hawkins et al., 2009; Keith et al., 2011);
3) the native intertidal *gastropod Patella depressa* with a 14 days larval duration (Deysher & Norton, 1982);
4) the common barnacle *Semibalanus balanoides,* a native intertidal species with a larval duration >30 days (Pineda et al., 2002);

These species were chosen as they are present on some of the structures within Poole Bay and have a variety of reproductive strategies and pelagic larval durations. *Styela clava* is a non-native species that is found on many subtidal structures in the region (Mallinson et al., 1999; Underhill-Day & P, 2005; Herbert et al., 2017). The gastropods *G. umbilicalis* and *P. depressa* are climate migrants and have been experienced range eastwards extensions in recent years in response to warming (Mieszkowska et al., 2006; Keith et al., 2011). The barnacle *S. balanoides* has a Boreal distribution and may retreat northwards with rising temperatures (Poloczanska et al., 2008; Jones et al., 2012).

Hydrodynamic model

In order to model the hydrodynamic conditions within Poole Bay, TELEMAC-2D (http://www.opentelemac.org/) was used to simulate the two-dimensional (2D) depth averaged flow. TELEMAC-2D allows a variable model resolution through the use of unstructured triangular mesh. The spatial resolution of the unstructured mesh varied between 10 m at the coastline and 500 m in open water (Figure 5.1). The TELEMAC hydrodynamic model was run over the period of 32 days to allow for two spring and two neap cycles and the output of which included velocity fields (*u* (*x, y*) and *v* (*x, y*)) and water depth at a series of time steps (every 10 seconds). The TELEMAC hydrodynamic model was calibrated using existing bathymetric and flow data of the area collected during the Channel Approach deepening studies (HR Wallingford, 2004) and AWAC current data from the Poole Bay near shore replenishment trial collected by Channel Coast Observatory (CCO, 2017).

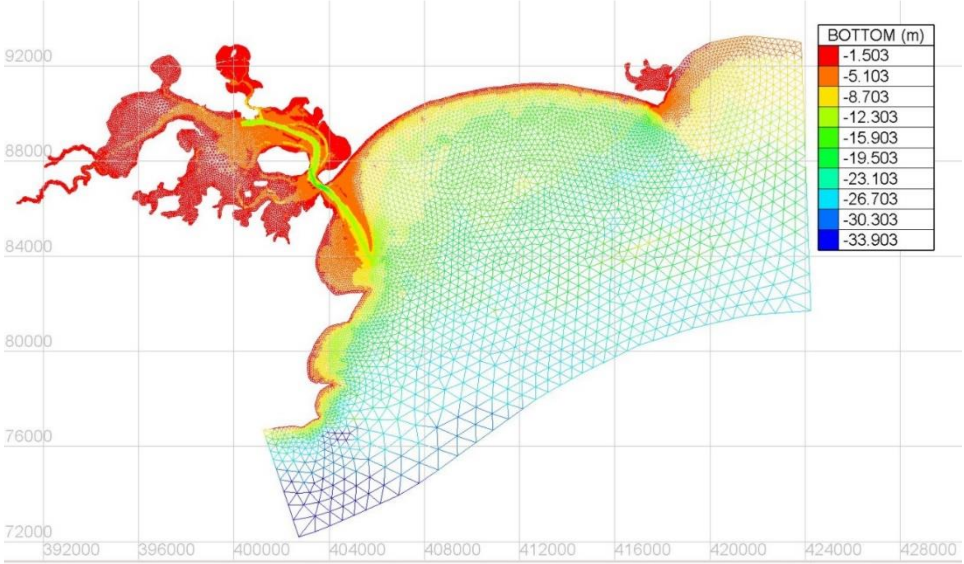


Figure 5.1 Triangulation mesh and bathymetry for Poole Bay. Figure created in Blue Kenue (https://www.nrc-cnrc.gc.ca/eng/solutions/advisory/blue_kenue_index.html)

Larvae behaviour model

Larvae particles were modelled using a Lagrangian larvae behaviour model created in Matlab R2016a (MathWorks, 2016). Larvae were assumed to be passive particles with no swimming capability that were transported around by the water currents at the point in which they are located at each time step. Each model scenario released a fixed amount of particles (2,000) from significant patches of known natural or artificial habitats, indicated as boxes (Figure 5.2), sites 3-9 were chosen to represent significant artificial structures within Poole Bay, including several piers (site 3,5,7) groynes (site 6,8,9) and Poole Port (site 4). Additional artificial structures are located within Poole Bay that were not included in the model. The larvae remained in the model for the pelagic larval duration of each species (24 hours, 6 days, 14 days and 30 days). Once they had reached their settlement state any larvae situated within a designated box were counted as a potential hit (Figure 5.2). The boxes in Figure 5.2 were designated based on the presence of hard substrate which included both natural and artificial shores. The number of larvae within each designated hit box were counted as a potential hit at each time step of the model from which connectivity matrices were constructed to identify pathways, barriers and stepping stones. Summary parameters are shown in Table 5.1 and additional information on these models can be seen in Appendix 3.

Table 5.1 Summary of set-up parameters for model.

Variable	**Value**	**Reference**
Number of particles	2,000	
Main time step	50s	
Sub-time step for vertical dispersion	5s	Mead (2008)
Minimum depth (dry)	0.1 m	
Coefficient of horizontal diffusion	0.02 m^2s^{-1}	Fischer *et al.* (1979)
Coefficient of vertical diffusion	0.001 m^2s^{-1}	Fischer *et al.* (1979)
Settlement rate	0.005 m^2s^{-1}	

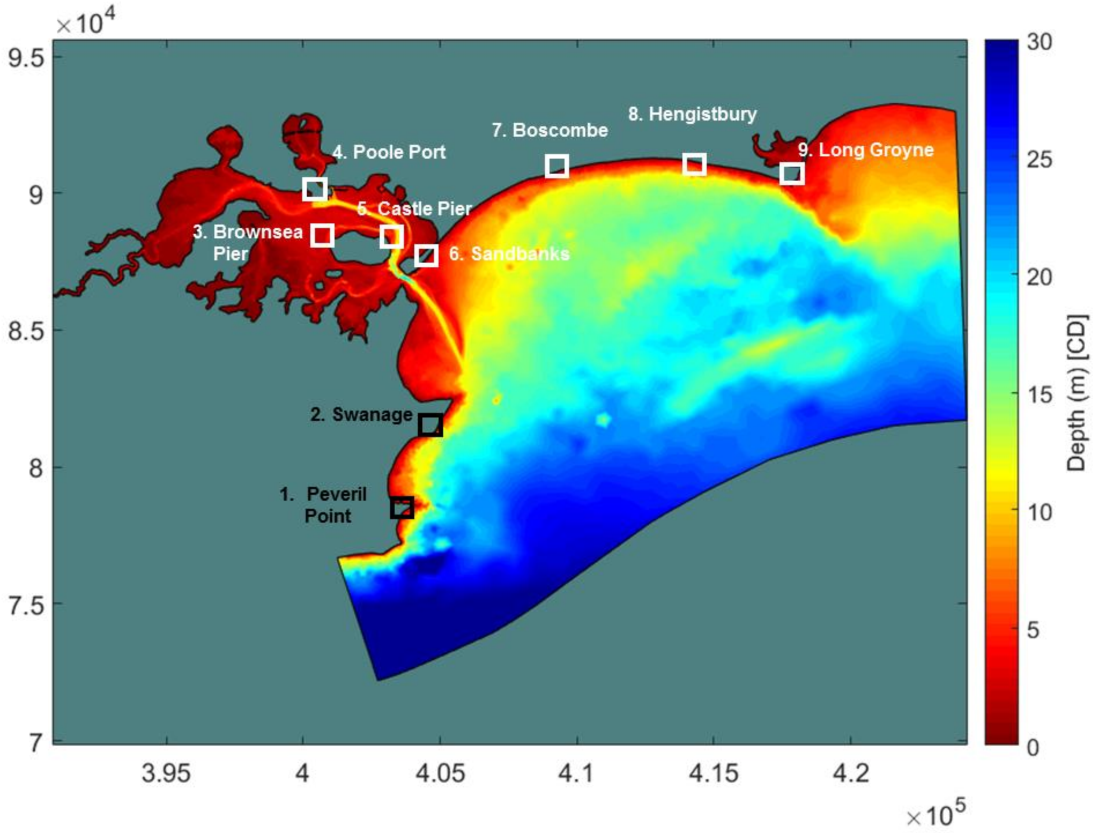


Figure 5.2 Source and destination locations within Poole Bay, Black boxes= Site 1 and 2 are natural rocky shores, White boxes= Sites 3-9 are artificial shores/structures. All artificial sites extend in to the intertidal. Sites 4 and 5 and 9 also include subtidal hard substrata.

5.2.3 Surveys of recruitment and adult distribution and model validation

The models were validated in the field survey to quantify species distribution of *S. clava, G. umbilicalis, P. depressa* and *S. balanoides* at Peveril Point, Swanage, Brownsea (Pottery) Pier, Poole Port, Brownsea Castle, Sandbanks, Boscombe, Hengistbury and Long Groyne were all surveyed during summer (June-August) 2016. The abundance of *P. depressa* and *S. balanoides* were based on density estimates collected using 0.25m^2 quadrats between MLW and MHW according to the Marine Nature Conservation Review (MNCR) SACFOR scale of abundance (JNCC, 2017). For *G. umbilicalis* and *S. clava* five 10 minute timed searches were conducted at each site and the abundance of individuals were recorded and converted to the SACFOR scale.

In addition the abundance of *S. clava* was included from existing subtidal surveys of the Boscombe Artificial Surf Reef (BASR) and Bournemouth and Boscombe Pier (Herbert et al., 2017 & Chapter 4)

5.3 Results

The models produced numerical data (Table 5.2, A3.1-A3.4), fixed figures (Figure 5.3, A3.1-A3.9) and video outputs to illustrate the response of larval particles to the hydrodynamic conditions within Poole Bay. There is a strong southerly current to the west of the model which created eddies around the headlands that send particles across Poole Bay to the east under the flood tide. The majority of the particles remain close to the coastline and the greatest number of particles lost from the model occurred at the southern model boundary south of Peveril Point.

Larvae released from the natural rocky shores at Peveril Point and Swanage were transported in both a southerly direction towards the model boundary but also a northerly direction towards Poole Harbour and after 24 hours, 6 days, 14 days and 30 days larvae were counted at various artificial shores/structures inside Poole Harbour (Figure 5.3a). Larvae released from Poole Port, a key introduction point for non-native species, are quickly transported to the natural shores within 24 hours (Figure 5.3b). As the entrance to Poole Harbour is so narrow there is a very fast flowing channel which drives larvae in and out of the harbour. The coastline around Poole Harbour and Swanage is very complex resulting in a variety of dispersal patterns.

The model indicates that particles released from artificial sites at Boscombe, Hengistbury and Long Groyne would not reach the two natural shores after 24hours- Swanage and Peveril Point (Figure 5.4 & 5.5). Yet particles with a PLD of 6-30 days released from the natural sites Peveril Point & Swanage can reach Boscombe, Hengistbury and Long Groyne, highlighting the dominant west to east hydrodynamic flow in the Bay.

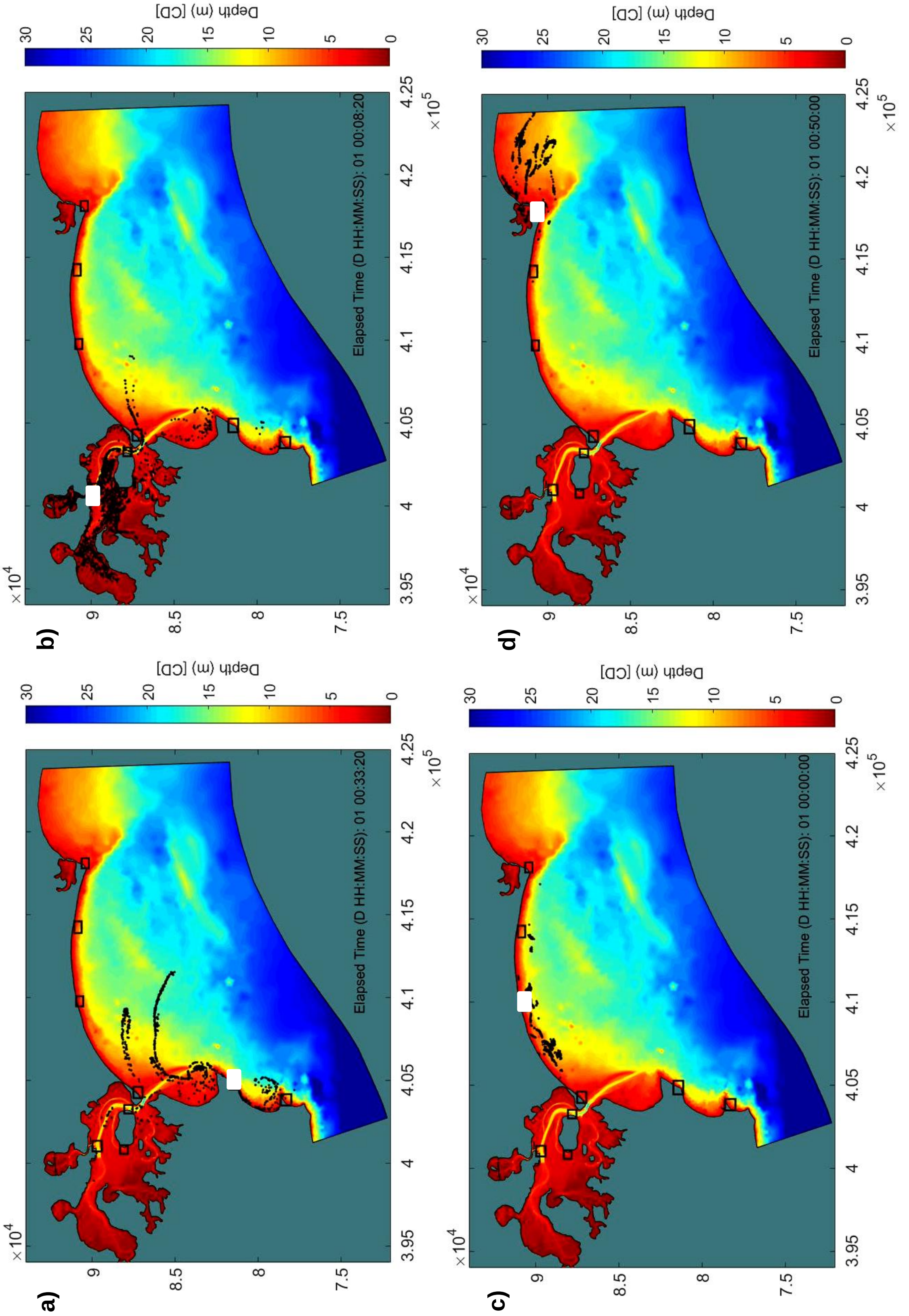


Figure 5.3 Model output showing location of particles after 24 hours released from a) Swanage, b) Poole Port, c) Boscombe d) Long Groyne (White box indicates release site, black dots represent particles/larvae, Black boxes indicate destination locations).

The levels of larval retention (number of particles which remained at the release site) at each site were low after 24 hours (Table 5.2), with the highest at Sandbanks (7.27%) and the lowest at Castle Pier (1.96%). After 30 days only two sites were still recorded to have larval retention which were Poole Port (0.10%) and Sandbanks (0.70%).

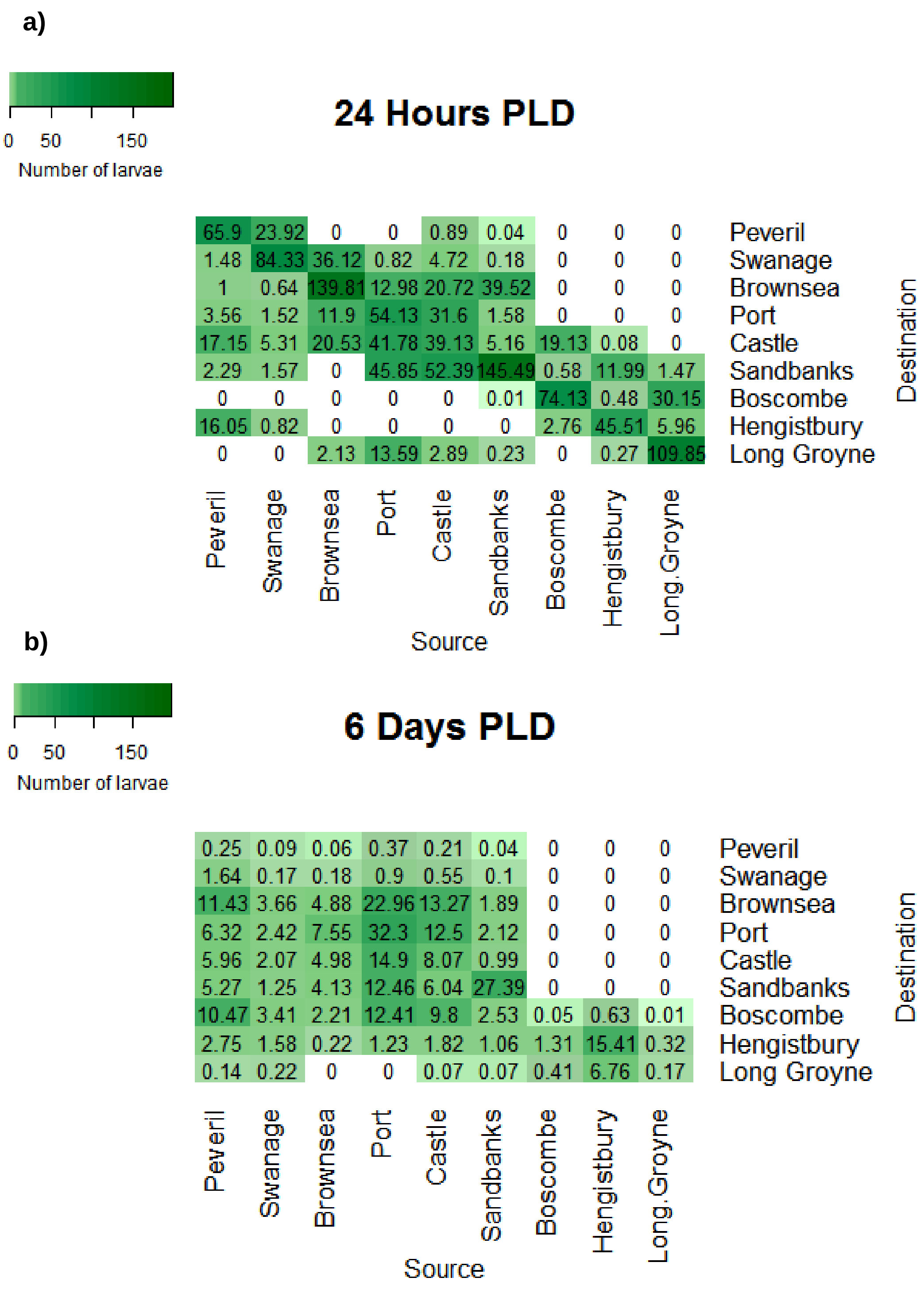
Table 5.2 Retention of larvae particles released from nine sources, over 24 hours, 7 days, 14 days and 30 days.

	Mean Proportion % Retention			
Source	**24 hours**	**6 days**	**14 days**	**30 days**
Peveril Point	3.30	0.00	0.00	0.00
Swanage	4.22	0.00	0.00	0.00
Brownsea (Pottery) Pier	6.99	0.00	0.13	0.00
Poole Port	2.71	0.02	0.54	0.10
Castle Pier	1.96	0.00	0.15	0.00
Sandbanks	7.27	0.01	1.39	0.70
Boscombe	3.71	0.00	0.01	0.00
Hengistbury	2.28	0.01	0.43	0.00
Long Groyne	5.49	0.00	0.00	0.00

The connectivity matrices show the degree of potential connectivity between sites after 24 hours, 6 days, 14 days and 30 days (Figure 5.4 & 5.5). A high proportion of the 2,000 particles are lost from the model within the first 24 hour period. The first hypothesis that larvae released from Poole Harbour cannot reach the natural shores has been rejected as within 24 hours the larvae released from the sites within Poole Harbour (Brownsea, Port, Castle) have the ability to reach the natural site at Swanage (Figure 5.4ai) and the larvae released from Castle has the ability to reach the second natural site at Peveril point.

The second hypothesis that larvae released from the natural shores at Peveril and Swanage will be able to reach the artificial shores within the bay is accepted, as within 24 hours the larvae released from the natural shores can reach seven out of the nine artificial shores in the bay (Figure 5.4ai) and within 6 days they can reach all nine artificial shores (Figure 5.4aii).

The third hypothesis that PLD will affect the distance travelled by larvae has also been accepted as species with a longer PLD have a greater dispersal distance within the model and therefore an increased chance of reaching a greater number of sites. Some sites are potentially connected after 24 hours which is important for species with low PLDs such as *S. clava*, however potential dispersal towards other sites take longer. For example, particles released from Peveril, Swanage, Brownsea (Pottery Pier), Poole Port, Castle and Sandbanks do not reach Boscombe until day 6 meaning a species with a shorter PLD such as *S. clava* would not be able to reach the shore at Boscombe yet *G. umbilicalis* would. In addition, it also takes 14 days for larvae released from Brownsea and Port to reach the furthest site east at Long Groyne, illustrating that species such as *P. depressa* have the ability to colonise this site. The sites to the east seem the least connected to the sites in the west as larvae released from Boscombe and Hengistbury with a 24 hours PLD have the ability to reach Sandbanks and Castle, yet after 24 hours there is no connectivity between Boscombe and Hengistbury to the east and all the remaining sites to the west. In addition, even after 30 days the larvae released from Boscombe, Hengistbury and Long groyne cannot reach the two natural shores at Peveril point and Swanage or the two artificial shores within the harbour Brownsea (Pottery Pier) and Poole Port.



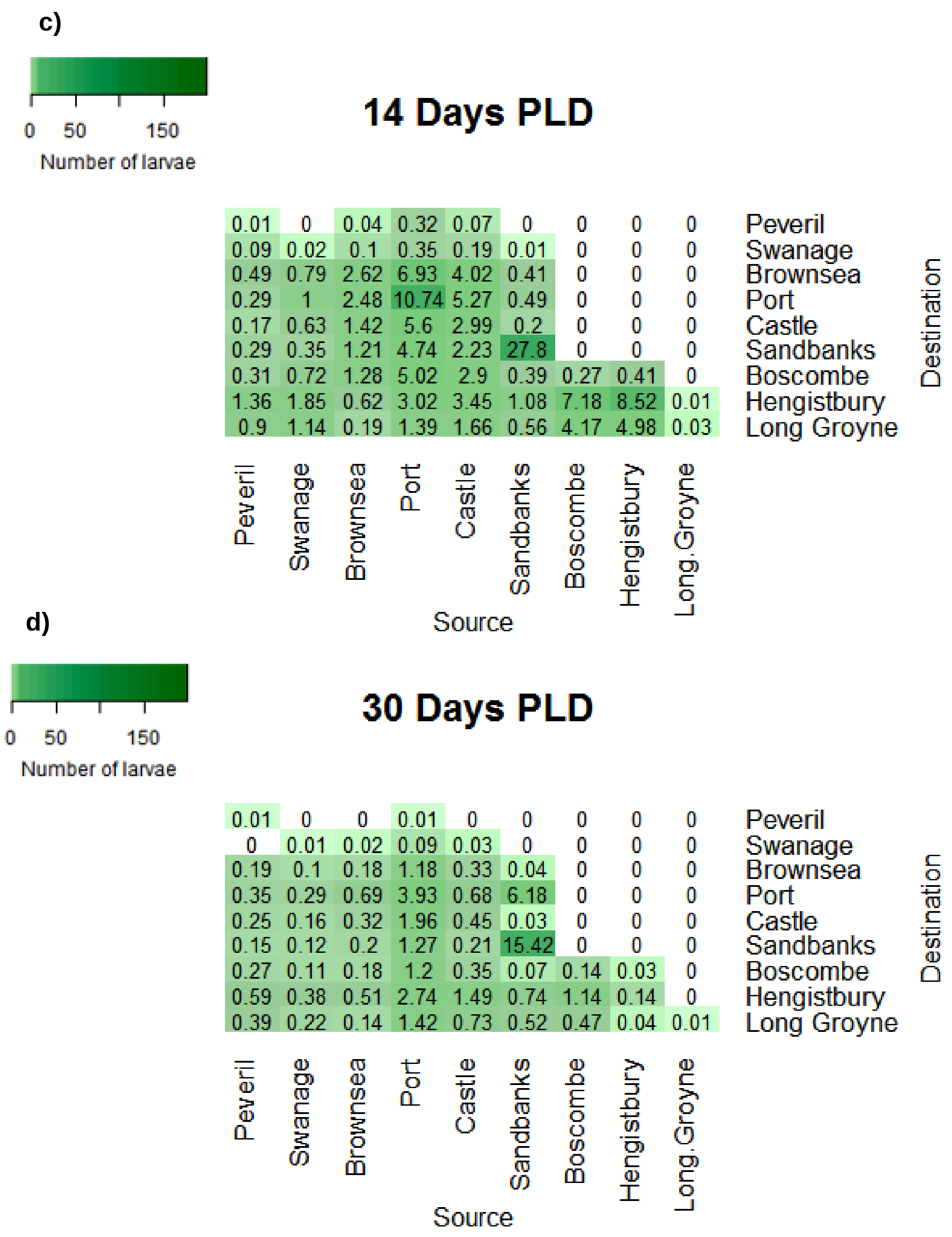
a) 24 Hours PLD

	Peveril	Swanage	Brownsea	Port	Castle	Sandbanks	Boscombe	Hengistbury	Long.Groyne
Peveril	65.9	23.92	0	0	0.89	0.04	0	0	0
Swanage	1.48	84.33	36.12	0.82	4.72	0.18	0	0	0
Brownsea	1	0.64	139.81	12.98	20.72	39.52	0	0	0
Port	3.56	1.52	11.9	54.13	31.6	1.58	0	0	0
Castle	17.15	5.31	20.53	41.78	39.13	5.16	19.13	0.08	0
Sandbanks	2.29	1.57	0	45.85	52.39	145.49	0.58	11.99	1.47
Boscombe	0	0	0	0	0	0.01	74.13	0.48	30.15
Hengistbury	16.05	0.82	0	0	0	0	2.76	45.51	5.96
Long Groyne	0	0	2.13	13.59	2.89	0.23	0	0.27	109.85

b) 6 Days PLD

	Peveril	Swanage	Brownsea	Port	Castle	Sandbanks	Boscombe	Hengistbury	Long.Groyne
Peveril	0.25	0.09	0.06	0.37	0.21	0.04	0	0	0
Swanage	1.64	0.17	0.18	0.9	0.55	0.1	0	0	0
Brownsea	11.43	3.66	4.88	22.96	13.27	1.89	0	0	0
Port	6.32	2.42	7.55	32.3	12.5	2.12	0	0	0
Castle	5.96	2.07	4.98	14.9	8.07	0.99	0	0	0
Sandbanks	5.27	1.25	4.13	12.46	6.04	27.39	0	0	0
Boscombe	10.47	3.41	2.21	12.41	9.8	2.53	0.05	0.63	0.01
Hengistbury	2.75	1.58	0.22	1.23	1.82	1.06	1.31	15.41	0.32
Long Groyne	0.14	0.22	0	0	0.07	0.07	0.41	6.76	0.17

Figure 5.4a Connectivity matrix and heat map for the number of larvae which reach each site with i) 24 hours PLD, ii) 6 days PLD from all nine source locations. (The numbers in the cells refer to the number of larvae).



14 Days PLD

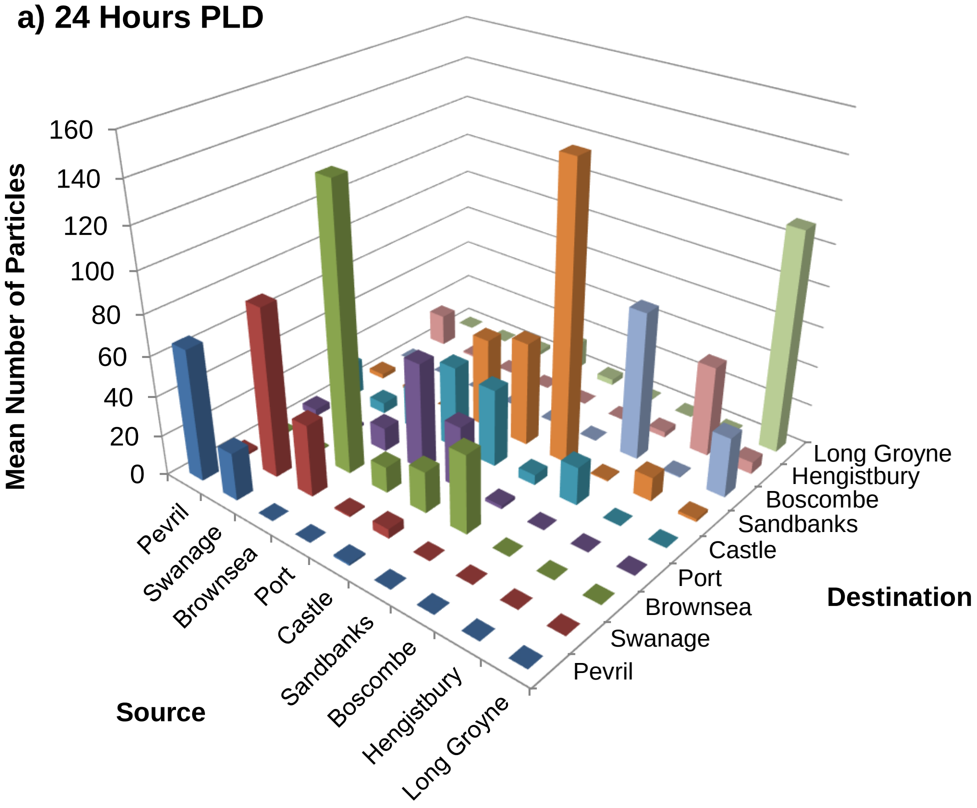
	Peveril	Swanage	Brownsea	Port	Castle	Sandbanks	Boscombe	Hengistbury	Long.Groyne
Peveril	0.01	0	0.04	0.32	0.07	0	0	0	0
Swanage	0.09	0.02	0.1	0.35	0.19	0.01	0	0	0
Brownsea	0.49	0.79	2.62	6.93	4.02	0.41	0	0	0
Port	0.29	1	2.48	10.74	5.27	0.49	0	0	0
Castle	0.17	0.63	1.42	5.6	2.99	0.2	0	0	0
Sandbanks	0.29	0.35	1.21	4.74	2.23	27.8	0	0	0
Boscombe	0.31	0.72	1.28	5.02	2.9	0.39	0.27	0.41	0
Hengistbury	1.36	1.85	0.62	3.02	3.45	1.08	7.18	8.52	0.01
Long Groyne	0.9	1.14	0.19	1.39	1.66	0.56	4.17	4.98	0.03

30 Days PLD

	Peveril	Swanage	Brownsea	Port	Castle	Sandbanks	Boscombe	Hengistbury	Long.Groyne
Peveril	0.01	0	0	0.01	0	0	0	0	0
Swanage	0	0.01	0.02	0.09	0.03	0	0	0	0
Brownsea	0.19	0.1	0.18	1.18	0.33	0.04	0	0	0
Port	0.35	0.29	0.69	3.93	0.68	6.18	0	0	0
Castle	0.25	0.16	0.32	1.96	0.45	0.03	0	0	0
Sandbanks	0.15	0.12	0.2	1.27	0.21	15.42	0	0	0
Boscombe	0.27	0.11	0.18	1.2	0.35	0.07	0.14	0.03	0
Hengistbury	0.59	0.38	0.51	2.74	1.49	0.74	1.14	0.14	0
Long Groyne	0.39	0.22	0.14	1.42	0.73	0.52	0.47	0.04	0.01

Figure 5.4b Connectivity matrix and heat map for the number of larvae which reach each site with iii) 14 days PLD, iv) 30 days PLD from all nine source locations. (The numbers in the cells refer to the number of larvae).

a) 24 Hours PLD



b) 6 days PLD

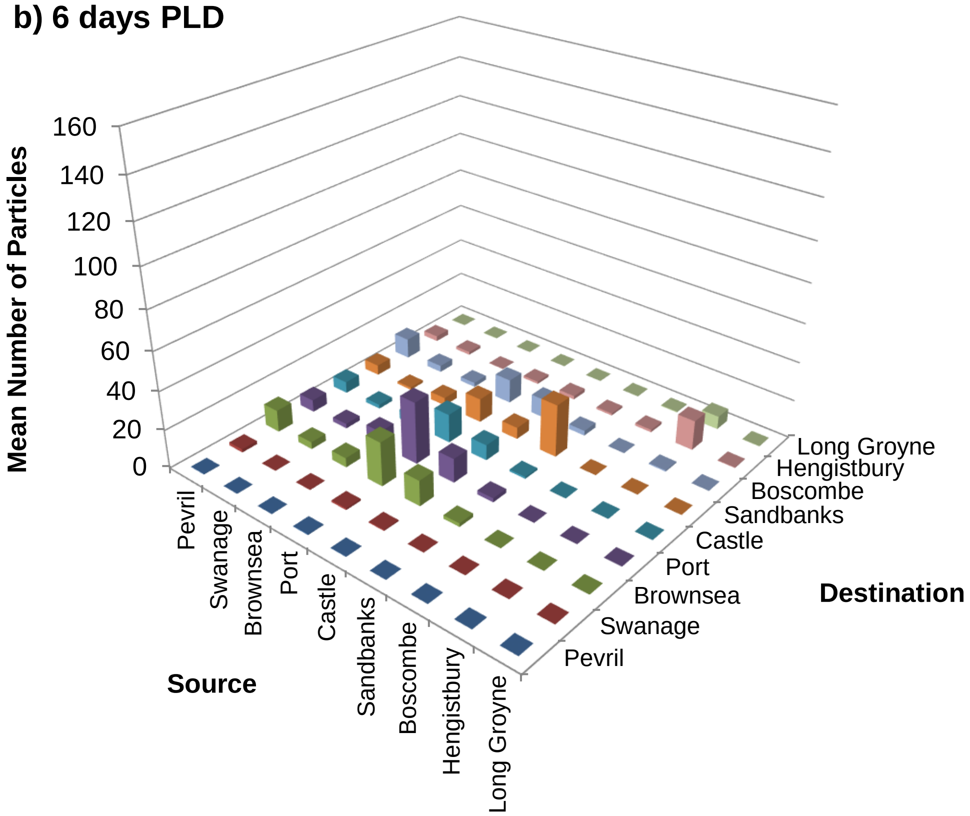
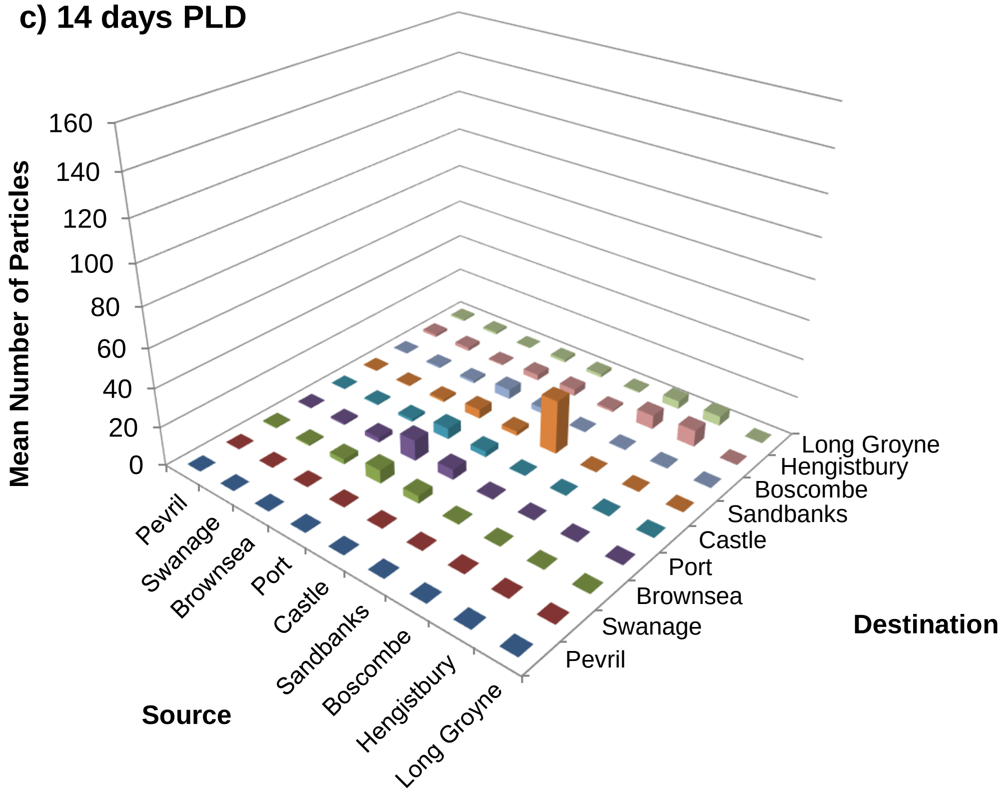


Figure 5.5a Dispersal matrices reporting mean number of larvae release from the source locations that then reached the destinations for i) 24 hours PLD (pelagic larval duration) ii) 6 days PLD.

c) 14 days PLD



d) 30 days PLD

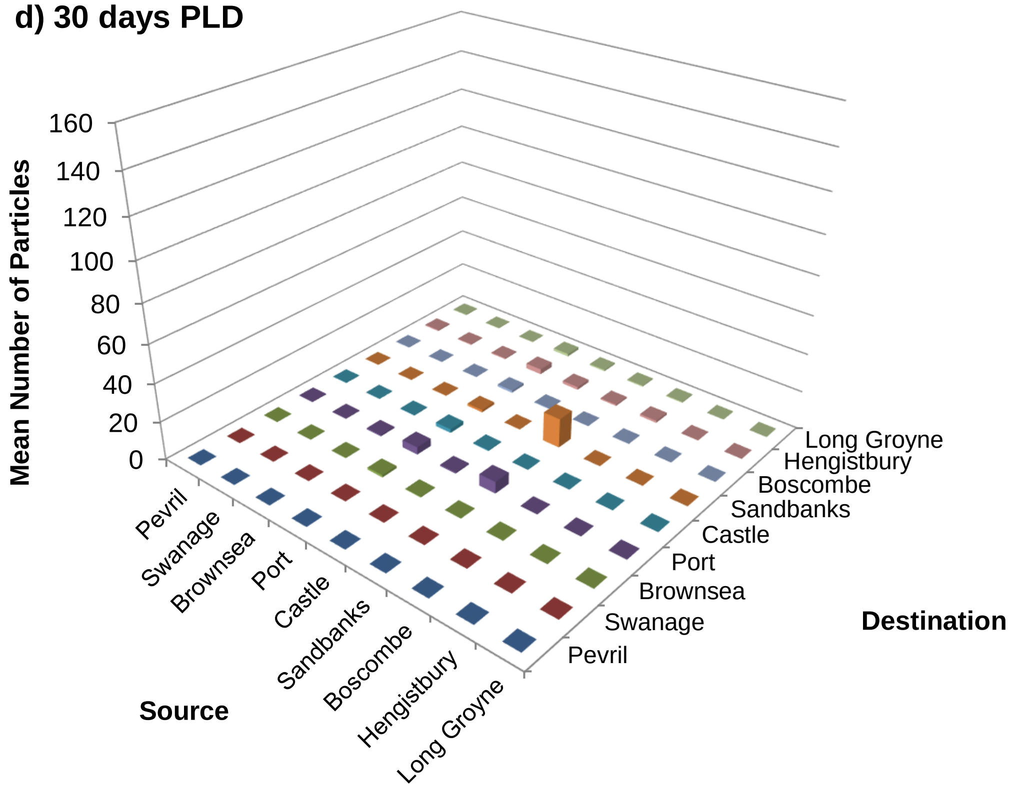


Figure 5.5b Dispersal matrices reporting mean number of larvae release from the source locations that then reached the destinations for iii) 14 day PLD (pelagic larval duration) iv) 30 day PLD

Model Validation

Within Poole Bay the abundance and distribution of *Styela clava, Gibbula umbilicalis, Patella depressa, Semibalanus balanoides* varied (Figure 5.6). Although the hydrodynamic model predicted that species with a 24 hour PLD such as *S. clava* could reach Peveril point and Swanage within 24hours they were only found inside Poole harbour and on the Boscombe Artificial Surf Reef. Yet the 6 day PLD hydrodynamic model for *G. umbilicalis* predicted that they would be found throughout Poole Bay, which field data showed. The hydrodynamic model for *P. depressa* indicated that after 14 days they could reach all sites within the bay, with greater connectivity to Boscombe and Hengistbury. The field data showed presence at these sites however they were found to be most abundant on the natural shores at Peveril and Swanage. The field data showed *S. balanoides* to be the most widely distributed and the most abundant species within Poole Bay, however the hydrodynamic model indicated quite low connectivity between sites after 30 days, particularly at the east of the bay (Figure 5.6).

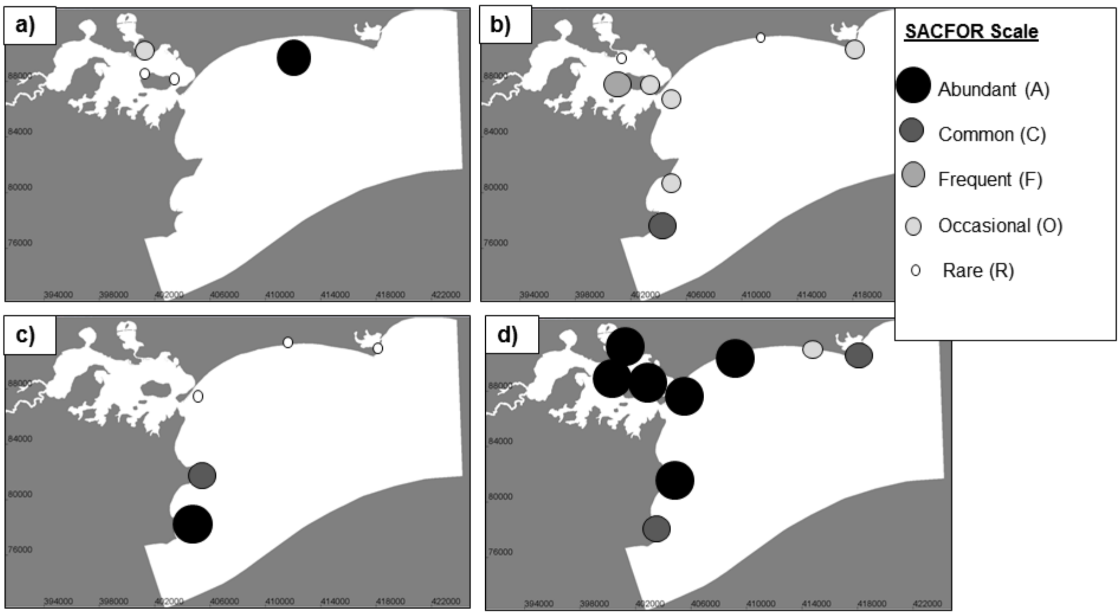


Figure 5.6 SACFOR abundance of a) *Styela clava*, b) *Gibbula umbilicalis*, c*) Patella depressa*, d) *Semibalanus balanoides* sampled within Poole Bay in June-August 2016. Additional subtidal records of populations of *S.clava* obtained from SCUBA and ROV surveys of Boscombe Artificial Surf Reef (Herbert et al., 2017).

5.4 Discussion

Poole Bay is a heavily managed coastline with artificial structures constructed for a variety of purposes including ports, tourism and coastal defences. The addition of artificial structures to the marine environment has the potential to alter the connectivity of metapopulations in the region. Connectivity of marine habitats is often overlooked by marine spatial planners even though areas may be protected by legislative designation. However the construction of new coastal infrastructure could have significant effects on the marine ecosystem.

By modelling the flow regimes of Poole Bay it has been possible to predict patterns in larvae dispersal between natural shores and artificial sites and structures. This research focused on the influence of tidal currents and did not include the effects of wind and waves; although previous studies have found these to be important factors they are difficult to model without adequate data (Pineda et al., 2007; Ayata et al., 2009). The benefit of focusing mainly on tides is they are relatively constant from year to year and within Poole Bay they create currents which as a general rule are roughly 10 times the speed of wind driven currents (Herbert et al., 2012). In contrast weather-related factors such as wind and waves are more highly variable. Rare extreme climatic events may also be important in determining the overall evolution of pelagic dispersal and therefore should be incorporated into future models.

Due to lack of species-specific information this model did not include larval behaviour but instead focused on generic pelagic larval durations which could be used to inform a wide range of species. Biological variability not modelled included aspects such as the buoyancy of a larva in tidal currents in the shallow waters, which can have profound impacts on larval dispersal. Furthermore, as mentioned above, PLDs are temperature-dependent and may be affected by other un-modelled factors such as nutrient availability, light levels (McCormick & Molony, 1995; Cowen & Sponaugle, 2009) and specific settlement behaviour such as parachuting, active swimming and other attachment related behaviours including chemical cues (Wethey, 1986; Kingsford et al., 2002; Sponaugle et al., 2002) and states of tide (Knights et al., 2006). However, although significant model refinements could undoubtedly be made by the incorporation of many of these factors, in a highly mixed, dynamic and complex open environment such as Poole Harbour and Poole Bay, it is most likely that the transport processes will be principally determined by tidal currents. Since larvae are difficult to follow in open water the PLD's are estimates based on laboratory studies, however their behaviour in the natural environment could be different and accelerated by prevailing temperature (O'Connor et al., 2007). Future developments of the model could also include a mortality model which would regulate the number of larvae arriving at a potential site and this would be determined by adult fecundity, temperature and food availability (Minchinton & Scheibling, 1991; Pineda, 2000). A settlement model was considered for this project however due to lack of species-specific information it could not be completed but future studies should aim to incorporate a settlement sub-model into the overall dispersal model.

Validation

Extensive work has been conducted on the validation of the physical model components (the hydrodynamic model and the Lagrangian particle model). The standard best practice methods were used for these models (Willis, 2011; Herbert et al., 2012) and the uncertainty in the accuracy of these physical components is likely to be orders of magnitude less than the uncertainty around the biological aspects of the species in question (Willis, 2011). The model accommodated the general uncertainty in the physical factors in the random walks which model variation below the spatial scale of the model. Nevertheless, wind and waves may have important heterogeneous physical implications for PLD and dispersal especially for larvae that spend time near the surface and these weren't modelled explicitly in this study but were accounted for using the random walks explained above. These models are classified as theoretical models in ecology (Schmitz, 2001) and their place in the scientific process is after initial data collection and during the formulation of new hypotheses that can be tested in the field. They are not designed to make accurate predictions of future events as with engineering models and consequently they are not generally calibrated or validated (Odenbaugh, 2005).

The hypothesis that larvae released from artificial sites in Poole Harbour would not reach the natural sites without assistance from a stepping stone in the form of an artificial structure was rejected at the scale of the current model as the strong hydrodynamic forces in the area resulted in good connectivity within 24 hours. This is vital to understand as any introduced species arriving in the port will have a strong potential to reach the closest natural rocky shores within 24 hours. Invasive non-native species such as *S. clava* are opportunistic invaders and have short PLD, therefore predicting their range is important in terms of future management and conservation especially with increased shipping movement in the region. *S. clava* is quick to colonise structures within ports and harbours and the increased addition of artificial structures will facilitate the dispersal of this species. Non-native species can cause significant economic damage to marine industries (Williams et al., 2010; Vila et al., 2014). Preventing non-native species introduction should be the main focus, however identifying potential pathways for dispersal is also important to monitor any impacts they might have on the natural environment (Sheehy & Vik, 2010).

The artificial sites acted as generational stepping stones for the 24 hour PLD as particles released from site 9 (Long Groyne) only reached as far west as site 6 (Sandbanks), yet when particles were then released from site 6 (Sandbanks) they were then able to be transported to sites 1-5. Without the stepping stone of Sandbanks there would be no direct hydrodynamic transport between the artificial sites in the east and the natural sites in the south. In order for Sandbanks to be a feasible stepping stone it would need to establish stable populations which could reproduce viable offspring (Wonham et al., 2000).

The larvae released from the natural sites can reach the artificial sites within Poole Harbour and the artificial site at the entrance to Poole Harbour (Sandbanks), which reinforces the connectivity between the natural and artificial sites. This also provides a potential explanation for how the short PLD of *G. umbilicalis* colonised the shore at Brownsea Pottery Pier shore and established a viable population. From the site within the harbour *G. umbilicalis* has then used the groynes at Boscombe as stepping stones over multiple generations for facilitating it range extension to the long groyne in the east of the region. Reinforcing the stepping stone hypothesis that without the presence of established populations on the structures, larvae released may not be able to reach these sites. The field observations partially support the findings. For example *P. depressa* is not commonly observed in the east of the region and the modelled PLD suggests they might be lost offshore when released from the natural sites, yet the presence of artificial structures within the bay may enable *P. depressa* to connect with sites to the east. The discrepancies for species with longer PLD such as *S. balanoides* could be due to several factors including larval behaviour and the influence of winds; the recruitment of barnacle cyprids is known to be correlated to strong winds particularly on complex coastlines (Hawkins & Hartnoll, 1982; Herbert et al., 2007). In addition a larger model may be needed to accurately model the dispersal of larvae with longer PLDs especially in a region with complex hydrodynamics such as Poole Bay. Shanks (2009) also stated that due to behavioural variation the pelagic larvae normally have shorter dispersal distances than that predicted by Lagrangian models which again highlights the need to conduct more in situ field work.

Pelagic larval duration (PLD) was shown to have an effect on the connectivity between sites as shown in previous studies (Shanks et al., 2003; Adams et al., 2014; Gormley et al., 2015). PLD is an important factor in a species reproductive strategy and clear differences were seen between the 24 hour PLD and the 30 day PLD. The greatest number of hits occurred within 24 hours whereas the majority of the larvae with a 30 day PLD had left the model domain by 30 days, suggesting a larger model is needed to assess the longer pelagic durations.

Although this dispersal model suggests that the artificial structures are acting a stepping stones it is difficult to come by hard evidence. The use of genetic markers could be used to assess the population linkages and connectivity between sites. Studying larval dispersal in the marine environment is very difficult and can be very expensive and time consuming but in order to improve our understanding, further studies will need to be conducted. The models described here help clarify and target such studies which are most important for future predictive investigations.

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6 Evaluation of Low Cost Ecological Enhancement Techniques to Improve Habitat Heterogeneity on Coastal Defence Structures

Abstract

Sea level rise and higher storm frequency are increasing the need for the placement of hard coastal defences worldwide. The majority of these defences lack optimal habitats for intertidal species, resulting in low diversity and abundance. Here, low-cost ecological enhancement techniques were evaluated on two different rock types that aimed to increase habitat heterogeneity and surface texture. Arrays of holes and grooves inspired by ‘blast features’ produced during the quarrying process were created on granite rock armour in the north of England at Runswick Bay, N. Yorkshire and limestone rock groynes in southern England at Poole Bay, Dorset. After 12 months, the treatments had attracted new species to the defence structures and increased the overall diversity and abundance of organisms compared to control areas. Mobile fauna including crabs and fish were also recorded utilising the holes and grooves in Poole Bay. Non-native species were recorded in grooves at one site however their abundance was not significantly different to that of control areas. At the southern site, species known to be spreading in response to climate change were found in treatments but not in control areas. The cost of the installation of these enhancement techniques was low in relation to that of the defence scheme and could be easily installed either during or after construction. Through evaluation of the use of these ecological enhancement techniques on coastal structures, it is suggested that they have considerable potential to enhance local patterns of biodiversity when used within large-scale coastal engineering defence projects.

6.1 Introduction

Hard coastal defence structures are predominantly constructed from materials that are different to the local geology and marine environment and are designed to be durable, effective and efficient (French, 2001; Dong, 2004). Coastal defence structures can form either a solid or permeable barrier, which can both absorb and dissipate wave energy, and are designed to provide a long-term cost effective way of protecting land or assets from flooding and erosion (French, 2001). A variety of materials including concrete, wood and rock are used, although placement of rock armour boulders has more recently been favoured due to the longevity and efficiency at dispersing wave energy (Bradbury & Allsop, 1987; Crossman et al., 2003). The type of rock used in a particular area can be determined by the cost of transportation and aesthetic influences, particularly in conservation areas. The design of coastal defence structures is then determined by the specific erosion risks and local environmental conditions (Crossman et al., 2003; Garcia et al., 2004).

Coastal defence structures are typically colonised by sessile species, such as algae, barnacles, mussels and hydroids (Moschella et al., 2005) with community composition differing due to the substrate type (Green et al., 2012), tidal height, orientation and location within a structure (Sherrard et al., 2016). The majority of structures lack surface heterogeneity and the ability to retain water at low tide (Bulleri & Chapman, 2004; Coombes et al., 2011; Firth et al., 2013b, 2016b). In comparison, natural shores have a variety of habitats including rock pools and crevices which provide a refuge from both biotic and abiotic pressures at all states of tide (Raffaelli & Hawkins, 1996; Little et al., 2009; Firth et al., 2013b). Additional factors which differ between natural and artificial structures are slope, shading, age, available space and disturbance levels (Glasby, 1999b; Knott et al., 2004; Moschella et al., 2005; Chapman & Blockley, 2009).

As mentioned in Chapter 1 adaptations can be made to coastal defence structures to encourage the colonisation and survival of intertidal species (Moschella et al., 2005;

Dyson & Yocom, 2015), a process termed ‘ecological enhancement’ or ‘ecological engineering’ (Mitsch, 2012; Firth et al., 2014b; Sella & Perkol-Finkel, 2015). The purpose of ecological enhancement is to increase and / or improve the habitat for biodiversity whilst also protecting human health and the environment (ITRC, 2004). These adaptations can take many forms, including features that can be retrofitted onto existing structures or be incorporated into the construction of new defence projects (Firth et al., 2014b, 2015, 2016b; Evans et al., 2015). When compared to natural shores, the main habitats that coastal defence structures usually lack are roughness, holes, crevices and areas of water retention such as pools (Moschella et al., 2005; Firth et al., 2013b). Evans et al., (2017) discovered that when considering multifunctional coastal defence structures, stakeholders considered the ecological benefits to be more important than the social, economic and technical benefits, which reinforces the application of ecological enhancement.

Trials have aimed to improve the habitat heterogeneity of artificial structures through increasing the texture of concrete (Coombes et al., 2015), drilling pits to seawalls (Martins et al., 2010), attaching precast concrete tiles (Borsje et al., 2011; Loke et al., 2015) in order to improve biodiversity. Small scale water-retaining features have also been trialled by omitting blocks in the concrete (Chapman & Blockley, 2009), attaching flowerpots to seawalls (Browne & Chapman, 2011) core drilling pools in rock armour (Evans et al., 2015) and moulding concrete between boulders to form pools (Firth et al., 2016a). All of these interventions have had a measure of success in increasing the variety of habitats on the structures, resulting in either an increase in species richness or a change in community composition (See Chapter 1, Table 1.1).

On a larger scale, pre-cast habitat enhancement units have been trialled that incorporate rock pools of varying sizes, crevices and pits (Firth et al., 2014b). Whilst these units can be incorporated into rock armour (Sella & Perkol-Finkel, 2015), it is difficult for them to be cheaply replicated, scaled-up and installed post-construction. This is important, as due to the prevalence of coastal defence structures, there is an outstanding need for low-cost retrofitting options, i.e. simple techniques which can be executed without large plant machinery or high construction costs.

The current study thus evaluates the application of low-cost ecological enhancement techniques on coastal defence structures in locations exposed to high wave energy. In such environments, the use of rock armour (2-20 tonne boulders) predominates and the placement of artificial pools or tiles on the boulders is not an option as these would be removed by wave action (Browne & Chapman, 2011). In addition, many of the structures are already in place without any ecological enhancement, and a technique that can be retrofitted post-construction would be useful. Inspired by holes and groove ‘blast features’, which have been observed to retain water on the surface of quarried blocks, the designs were created to be replicated on any defence structure, including groynes, breakwaters and rock armour. These trials aimed to determine if these ecological enhancement techniques (“holes” and “grooves”) resulted in differences in community composition, species richness, total abundance, and species diversity when compared to non-manipulated (control) rock faces.

The following hypotheses were tested:

1) Species richness, total abundance and species diversity will be greater in the test areas than on the control un-manipulated rock.

2) The community composition will vary between test and control areas.

3) There will be significantly more water retention in the test areas compared with the controls.

4) There will be an increased number of habitat-forming functional groups (barnacles) and grazers (limpets) in the test areas compared to the controls.

6.2 Methods

6.2.1 Study Sites

Field trials were conducted to examine the ecological response of rocky shore species to two different enhancement treatments at each of two sites: Runswick Bay, North Yorkshire and Boscombe in Poole Bay, Dorset (Figure 6.1a). Runswick Bay is a popular tourist area with a moderately exposed sandy shore and shale bedrock platforms approximately 100 m to the north of the test site. The rock armour consists of 5-10 tonne granite boulders (sourced from the High Force Quarry in Middleton, UK), and was constructed in 2000 to dissipate wave energy and reduce overtopping of defences. Poole Bay is moderately exposed, has an urbanised coastline and is a popular tourist destination. It is predominantly sandy and the test site at Boscombe experiences a prevailing eastward longshore drift. Intertidal rock comprised of sandstone boulders is located 11 km west at Swanage. The test site had 3-6 tonne Portland limestone rock armour which was constructed in 2010 at Mean Low Water to strengthen the toe of older concrete groynes. Compared to nearby natural shores the rock armour at both study sites had a low abundance and diversity of colonising species, yet included barnacles and limpets that are important constituents of rocky shore ecosystems (Raffaelli & Hawkins, 1996). Prior to commencement of trials, Runswick Bay boulders supported a low diversity of species and small numbers of barnacles, limpets and intertidal snails, whereas Poole Bay had a more diverse community and supported greater numbers of barnacles, limpets, mussels and filamentous green, red and brown algae.

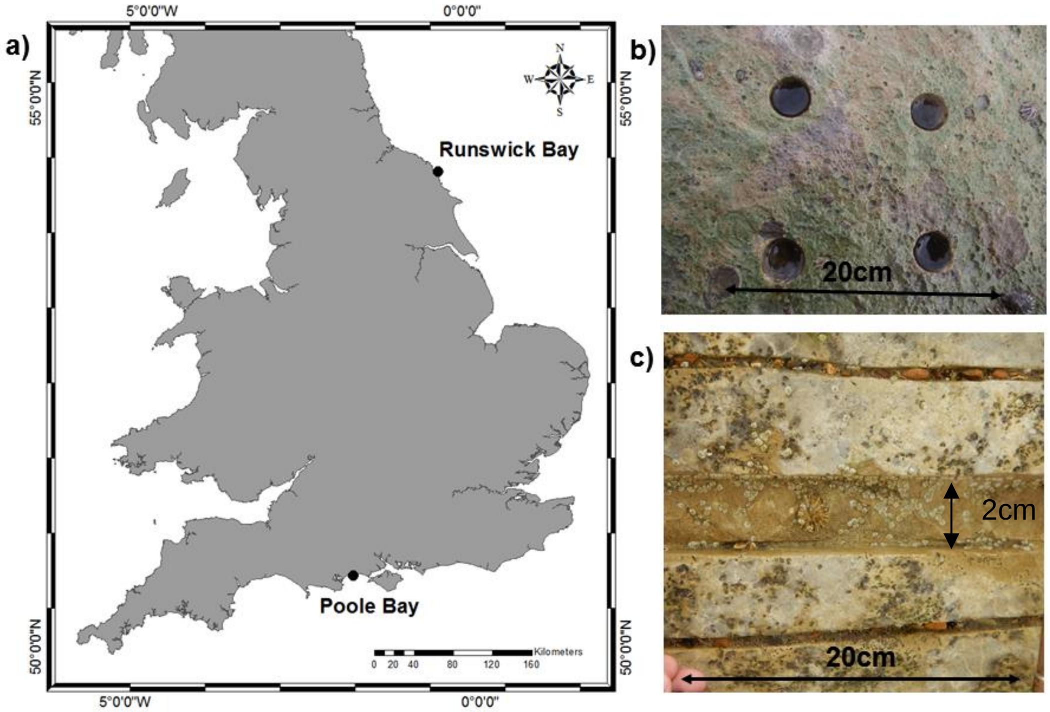

Figure 6.1 Site locations of Poole Bay and Runswick Bay with examples of (a) an array of Holes and (b) an array of Grooves.

6.2.2 Experimental design

Where logistically possible, treatments were created on the centre of the outer surface of separate boulders. Prior to creation of the treatments, flora and fauna were removed from each area with a paint scraper and wire brush. Two different enhancement treatments were evaluated at both sites.

(a) ‘Holes’, consisting of an array of four 20 mm deep x 16 mm diameter holes, orientated to retain water at low tide, were drilled into vertical and horizontal surfaces of boulders using a hand drill.

(b) ‘Grooves’ aimed to replicate the groove-microhabitat seen on natural rocky shore and occasionally observed in rock armour as a consequence of use of explosives in the quarrying process. Each array consisted of two, thin horizontal grooves (approx. 60 cm long x 0.3 cm deep x 1cm wide) and one thicker, coarser groove (approx. 60 cm long x 1cm deep x 2 cm wide) that were cut in to the rock using a petrol saw/angle grinder. The coarser middle grooves were chiselled out, which created a rough surface texture on the base and sides of the groove (Figure 6.1c).

At Runswick Bay, two arrays of holes were created on each of six separate boulders (N=12) (Figure 6.1b, Figure 6.2a). In addition, three arrays of grooves were created on a further six separate boulders (N=18) (Figure 6.1c, Figure 6.2b). All boulders were located between Mean Tide Level (MTL) and Mean Low Water at (MLW).

At Poole Bay a larger trial was conducted in which two arrays of holes were created on twenty-four boulders located across two rock groynes (N=48) (Figure 6.1b, Figure 6.2c). In addition, three arrays of grooves were created on twenty-four separate boulders located across two groynes (N=72) (Figure 6.1c, Figure 6.2d).

At both sites, equal numbers of fixed 20 x 20 cm control areas were created near each treatment on the same boulders by removing encrusting fauna and flora with a wire brush and paint-scraper to create a bare surface.

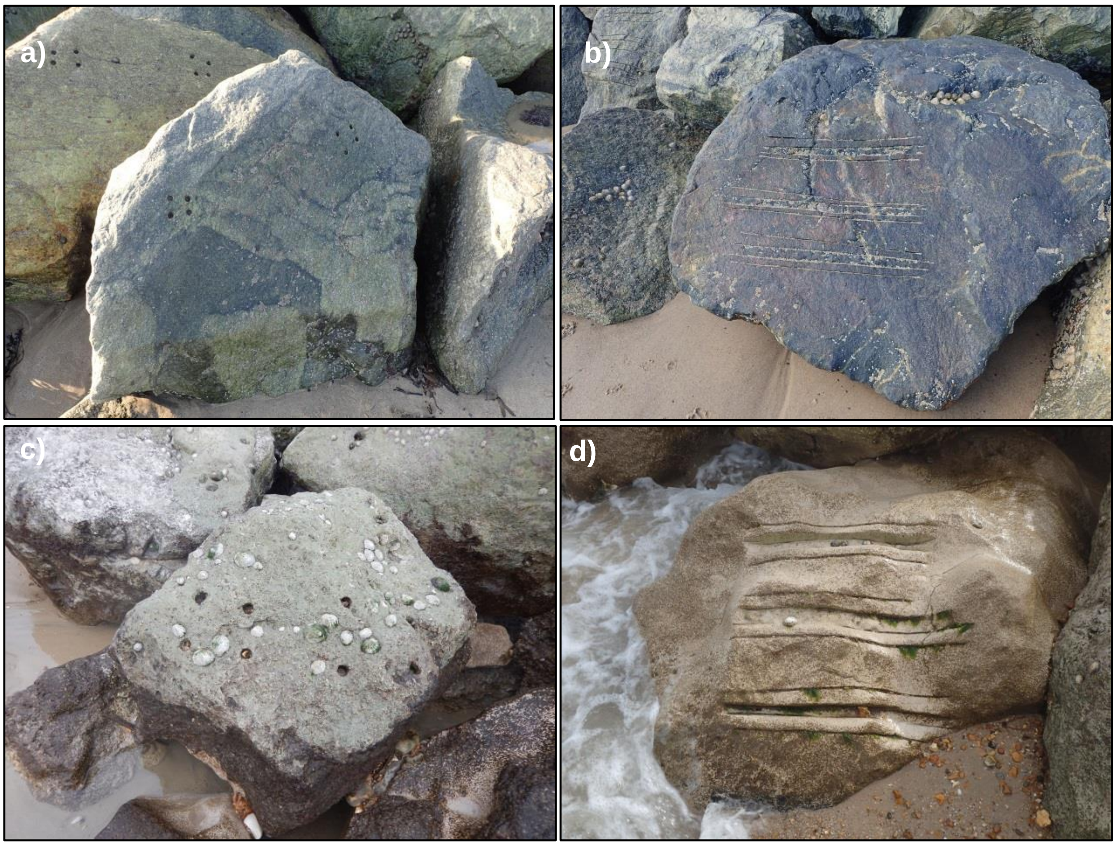

Figure 6.2 Positioning of a) Holes at Runswick, b) Grooves at Runswick, c) Holes at Poole Bay, d) Grooves at Poole Bay.

The financial cost of the treatments in Poole Bay was £500 which covered the cost of two workers' wages for 4 hours and the cost of a replacement blade/ drill bit. For Runswick Bay the cost was £660. As the Runswick Bay structures were built of granite, the time taken to complete the enhancements was longer than in Poole Bay due to the hardness of the rock, so less replication of treatments were made. In addition diamond tipped drill bits and blades were needed at Runswick Bay which were included in the overall cost.

6.2.3 Surveillance

At both sites, all boulders were sampled prior to installation of treatments to obtain base-line data on species abundance. Treatments and controls were established in

October 2014 at Runswick Bay and March 2015 at Poole, and then sampled after 12 months. The boulders with holes were sampled using a 20 x 20 cm quadrat placed over each array and control areas and the percentage cover of seaweed, diatoms and mussels and counts of fauna such as barnacles, limpets and snails were recorded to measure species abundance.

For boulders with grooves, three 20 x 20 cm quadrats were placed on each array and within adjacent control areas. Percentage cover of water retention and sediment in each treatment and control quadrat was also recorded. Water retention consisted of a pool of water and did not include damp areas. Photographs of all quadrats were taken to illustrate changes in communities. An estimate of surface heterogeneity of the rocks (in order to account for the increased surface area due to treatments) in each sampled quadrat was made using a fine scale variation of the chain and transect method (Luckhurst & Luckhurst, 1978). A thin chain was secured at the top of the quadrat and run to the bottom edge ensuring it touched the bedrock. This distance was then measured and used as a measure of relative surface texture (space available for colonisation (Loke & Todd, 2016)) within each quadrat sampled.

6.2.4 Statistical Analysis

In order to account for the increased surface area provided through the installation of holes and grooves onto a boulder surface, a correction factor was applied to standardise all abundance data collected from treatment quadrats. This was calculated using an average of the surface area measurements collected across all quadrats for each treatment, the correction factor applied to abundance data was 0.8 for quadrats containing grooves and 0.82 for quadrats containing holes.

Species richness, total abundance and Shannon-Weiner species diversity were determined using the DIVERSE function in PRIMER-e V6 (Clarke, 2001). A one-way

ANOVA with Whites adjustment to account for heteroscedasticity was performed for each site separately with treatment as the main factor (Long & Ervin, 2000). Any significant effects were explored using a Tukey post hoc test. A Bray Curtis similarity matrix was generated from square-root transformed data and the ANOSIM procedure used to test if there was any significant difference in communities of benthic organisms between treatments (Clarke, 2001). The SIMPER routine was used to determine species contributing most to the similarity within treatments and dissimilarity between treatments and controls (Clarke, 2001).

To determine if there was a difference in the average number of barnacles and limpets recorded in the different treatments verses the control areas, a negative binomial Generalised Liner Model (GLM) was applied. Due to numerous zero observations in count data this model resolved issues relating to over-dispersion and had the lowest Akaike Information Criterion (AIC) of the models trialled and, after examination of the residuals, was determined to be the most applicable to the data (Zuur et al., 2009). All analysis was undertaken in R using the MASS routine (Venables & Ripley, 2002) and base package (R Core Team, 2016).

6.3 Results

6.3.1 Runswick Bay – granite rock armour

On the granite rock armour boulders at Runswick Bay, only 3 species were recorded in the control areas after 12 months (Table 6.1), whilst an additional 5 species were observed to have colonised the holes and additional 10 species in the grooves. These new species included algae *Porphyra* sp., *Fucus* sp. and *Mastocarpus stellatus*, two gastropod snail species *Littorina saxatilis* and *Melarhaphe neritoides* and the mussel *Mytilus edulis* (Table 6.1).

Table 6.1 Presence and absence of species after a 12 month period for the holes, grooves and controls at Runswick Bay and Poole Bay (* indicates presence).

Group	**Species**	**Runswick Bay**				**Poole Bay**			
		Before	Holes	Grooves	Control	Before	Holes	Grooves	Control
Algae	*Ceramium* sp.						*	*	*
	Chaetomorpha sp.							*	
	Cladophora rupestris							*	
	Codium fragile							*	*
	Diatom					*	*	*	*
	Dumontia cortorta							*	
	Fucus sp.		*	*					
	Halurus sp							*	*
	Lomentaria articulata							*	*
	Mastocarpus stellatus			*					
	Polysiphonia sp.						*	*	
	Porphyra sp		*	*			*	*	
	Rhodochorton purpureum			*					
	Rhodothamniella floridula			*			*	*	*
	Scytosiphon lomentaria						*	*	*
	Ulva lactuca			*		*		*	*
	Ulva linza		*	*	*	*	*	*	*
Cnidaria	*Actina equina*							*	*
	Anemonia viridis							*	
Annelida	*Polydora ciliata*						*	*	
Crustacean	*Austrominius modestus*							*	*
	Perforatus perforatus						*	*	
	Carcinus maenas						*	*	
	Idotea granulosa			*					
	Semibalanus balanoides	*	*	*	*	*	*	*	*
Mollusca	*Lepidochitona cinereus*						*	*	
	Littorina saxatilis		*	*					
	Melarhaphe neritoides		*	*					
	Mytilus edulis		*	*		*	*	*	*
	Nucella lapillus						*	*	
	Patella depressa								*
	Patella vulgata	*	*	*	*	*	*	*	*
	Rissoa sp.							*	*
Bryozoa	Bryozoa sp.						*	*	
Ascidicea	*Ascidiella aspersa*						*	*	
Chordata	*Lipophrys pholis*						*		
Total Number of Species		**2**	**8**	**13**	**3**	**6**	**18**	**28**	**16**

There was a highly significant increase in species richness, species diversity and total abundance of individuals in both the holes (Table 6.2a; Figure 6.3a) and grooves treatments (Table 6.3a; Figure 6.2b) compared to the controls (P<0.001) at Runswick Bay. Both treatments created novel areas of water retention which were lacking on the control sites (Figure 6.4).

Table 6.2 Results of ANOVA with White's adjustment for comparison in species richness, total abundance and species diversity in holes and control and grooves and control at a) Runswick Bay and b) Poole Bay after 12 months (***=P<0.001, **=P<0.01,*=0.05 NS= Not significant).

a) Runswick Bay

	Species richness			**Total abundance**			**Species diversity**		
Source of Variation	*df*	F	p	*df*	F	p	*df*	F	p
Holes & Control	1	32.99	***	1	25.82	***	1	31.42	***
Grooves & Control	1	294.52	***	1	149.60	***	1	203.44	***
Contrasts	*df*	t	p	*df*	t	p	*df*	t	p
Holes- Control	130	-4.911	***	130	-4.15	***	130	-4.649	***
Grooves- Control	376	-13.86	***	376	-8.78	***	376	-10.88	***

b) Poole Bay

	Species richness			**Total abundance**			**Species diversity**		
Source of Variation	*df*	F	p	*df*	F	p	*df*	F	p
Holes & Control	1	10.25	**	1	2.77	*	1	13.70	***
Grooves & Control	1	127.73	***	1	0.28	NS	1	76.65	***
Contrasts	*df*	t	p	*df*	t	p	*df*	t	p
Holes- Control	166	-2.47	**	166	2.78	**	166	-2.61	***
Grooves- Control	1078	-10.46	***	1078	0.64	NS	1078	-8.3	***

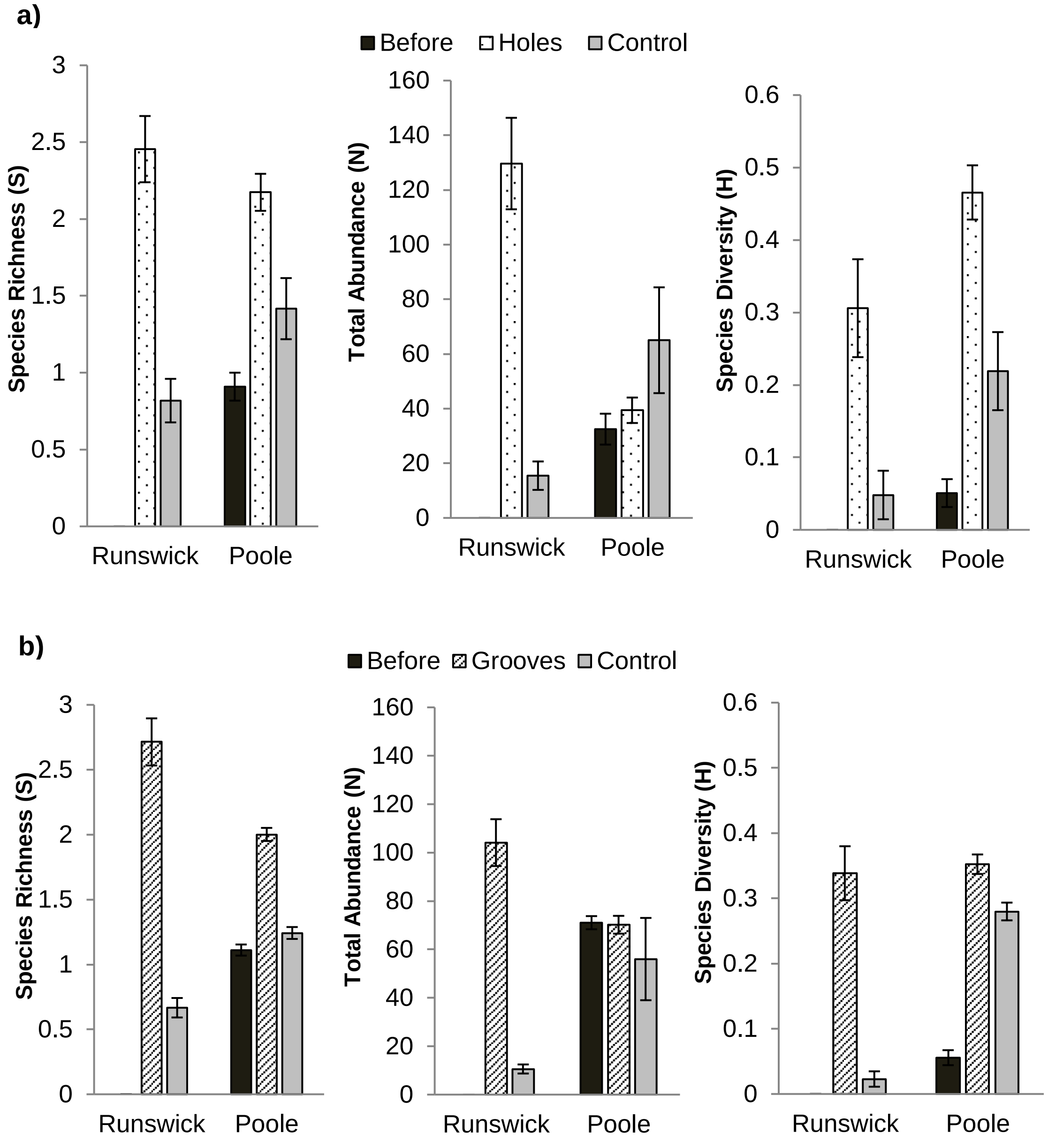

Figure 6.3 Mean species richness (S), total abundance (N) and species diversity (H) for a) holes and b) grooves before installation compared to the test and control after 12 months at Runswick Bay and Poole Bay (+/- SE).

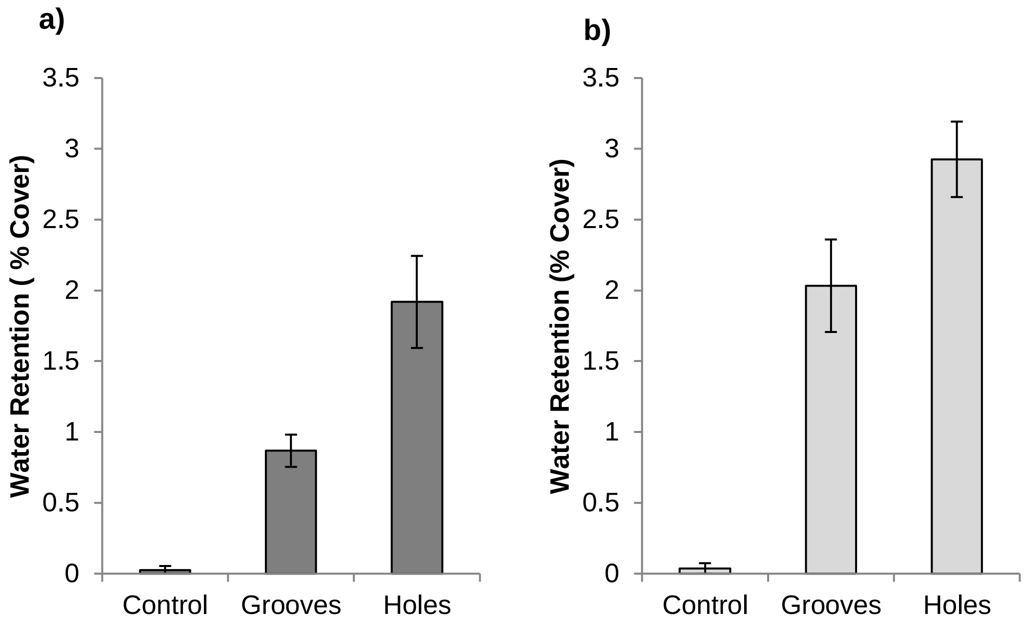

Figure 6.4 Mean percentage of water retention for the control, holes and grooves at a) Runswick Bay and b) Poole Bay (Mean +/- S.E.).

Community structure was found to be significantly different between the holes and grooves in comparison to the control areas (ANOSIM, R=0.18, P<0.01). The species contributing to 80% of the dissimilarity between holes and controls were *S. balanoides* (56% contribution to dissimilarity) and *U. linza* (21% contribution), with *S. balanoides* (Mean % cover, Holes=7.91, SE=1.24, Control =2.20, SE=0.26) and *U. linza* being most abundant in the holes (Mean % cover, Holes=2.59 SE=0.91, Controls=0.14 SE=0.11). The 75% dissimilarity found between the grooves and controls were due to a higher abundance of *S. balanoides* and *L. saxatilis* in the grooves.

There were significantly higher counts of habitat forming barnacles in both the grooves and holes treatments at Runswick Bay compared to the controls (Table 6.3a & Figure 6.5). A different pattern was seen with limpet abundance, with more occurring in the holes than either grooves or the control, however this was not significant (Table 6.3b & Figure 6.5b).

Table 6.3 Summary of the results of the negative binomial GLM applied to a) barnacle and b) limpet count data with treatment as the factor at a) Runswick Bay b) Poole Bay (***=P<0.001, **=P<0.01,*=0.05 NS= Not significant).

a) Runswick Bay

i) Barnacles			**AIC=1466.6, Theta=0.511**	
	Estimate	**Std. Error**	**Z value**	**P value**
Intercept	2.446	0.159	15.402	***
Grooves	2.074	0.237	8.730	***
Holes	1.909	0.339	5.637	***
ii) Limpets			**AIC=199.68, Theta=0.175**	
Intercept	-2.650	0.341	-7.753	***
Grooves	0.076	0.516	0.148	NS
Holes	0.840	0.620	1.354	NS

b) Poole Bay

i) Barnacles			**AIC= 1476.2 Theta= 0.043**	
	Estimate	**Std. Error**	**Z value**	**P value**
Intercept	1.744	0.294	5.921	***
Grooves	1.003	0.438	2.287	**
Holes	-1.271	0.757	-1.678	NS
ii) Limpets			**AIC=3007.5, Theta=0.122**	
Intercept	0.059	0.141	0.423	NS
Grooves	0.059	0.184	1.326	NS
Holes	1.213	0.280	4.333	***

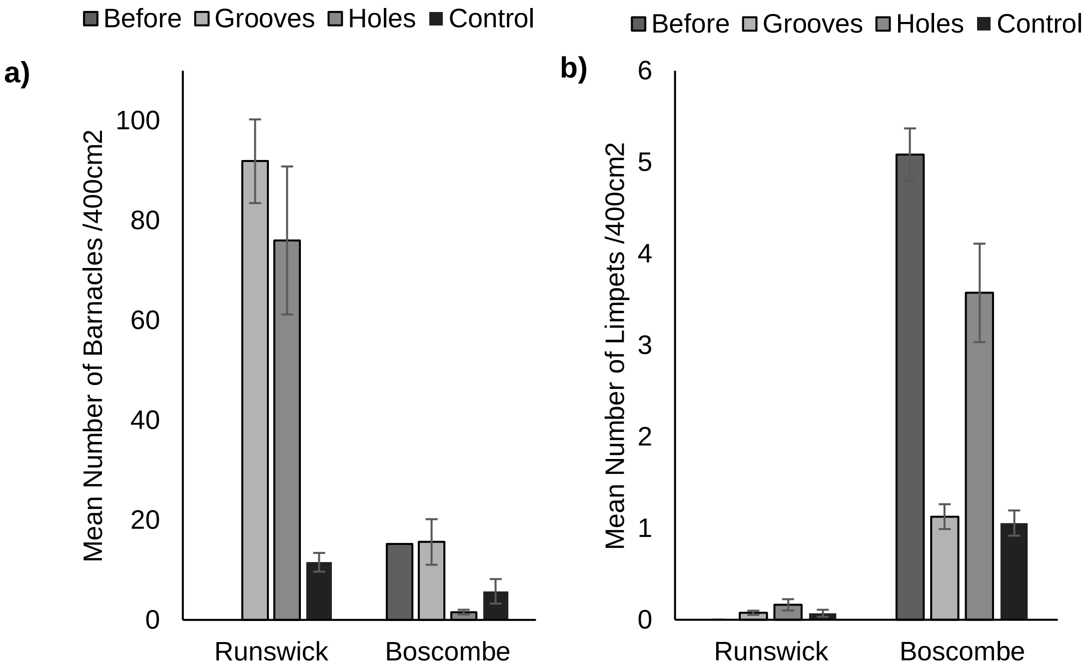

Figure 6.5 Mean abundance of a) Barnacles and b) Limpets in the control, grooves and holes quadrats at Runswick Bay and Poole Bay (Count data, Mean +/- S.E).

6.3.2 Poole Bay– limestone rock armour

The rock groynes at Poole Bay had higher baseline species richness compared with Runswick Bay, with 18 taxa found on the cleared control areas (Table 6.3). More taxa were located in the grooves treatments (28 taxa) than the holes (18 taxa). Species that were only found within the holes and groove treatments included *Ascidiella aspersa, Anemonia viridis, Carcinus maenas* and bryozoans (Table 6.1). Overall, species richness and diversity were both significantly higher in the holes and grooves treatments than that found on the control areas (P < 0.01; Figure 6.3, Table 6.2b). However, there was no significant difference in total abundance between the grooves treatment and controls, yet a significantly higher abundance was observed in the holes (P < 0.1; Figure 6.3, Table 6.2b). Consistent with Runswick Bay, the grooves in Poole Bay resulted in the greatest increase in species diversity. Two non-native species were recorded at Poole Bay; *Codium fragile* was recorded on the holes, grooves and control quadrats whereas *Austrominius modestus* was only recorded in the control and grooves quadrats.

ANOSIM showed a significant difference in community similarity between treatments (ANOSIM, R=0.026, P<0.001). Species contributing greatest to 87% of the dissimilarity between holes and control were *Rhodothamniella floridula* (26% of the dissimilarity), diatoms (24%) and *Patella vulgata* (23%) with *R. floridula* (Mean % (SE) cover, Holes=2.49 (0.48), Control=1.63(0.21)) and *P. vulgata* being most abundant in the holes (Mean abundance Holes=2.03 (0.31), Controls=0.49 (0.06)). Diatoms were most abundant on the controls (Mean abundance, Control=3.12 (0.42), Holes=0.50 (0.27)). The 81% dissimilarity between the grooves and controls were primarily the result of higher abundances of diatoms (Mean Abundance Grooves=3.59 (0.27), Control= 3.12(0.42)), *R. floridula* (Mean Abundance, Grooves = 1.76 (0.22), Control = 1.63 (0.21)), *S. balanoides* (Mean Abundance Grooves = 1.68 (0.27), Control= 0.51 (0.12)), and *P. vulgata* (Mean Abundance Grooves = 0.67 (0.08)), Control = 0.49 (0.06) in the grooves compared to controls. Although differences were small, water retention was higher in the holes (2%) and grooves (1%) compared with the control quadrats (0%) (Figure 6.4).

There were a significantly higher number of barnacles found in the grooves than in the control and holes treatment quadrats (Table 6.3b, Figure 6.5). However, the number of limpets was significantly higher in the holes treatment compared to the control and grooves samples.

6.4 Discussion

At the treatment scale the holes and grooves ecological enhancement techniques on both the granite rock armour at Runswick Bay and the limestone rock groynes in Poole Bay significantly increased species richness and diversity compared to the un-manipulated control areas (Tables 6.2 and 6.4) . The creation of holes on the boulders also significantly increased total abundance of organisms on both shores (Tables 6.2 and 6.4), whereas total abundance in the grooves treatment was only significant for the granite boulders at Runswick Bay. The type of rock used to construct coastal defence structures has been shown to affect community composition, with hard, fine-grained rocks, such as basalts, supporting less diverse communities than sandstones (Green et al., 2012) and limestones (Sherrard et al., 2016). During this study, greater species richness overall (Table 6.1) was observed on the limestone boulders at Poole than on the granite boulders at Runswick Bay. Biogeographic factors may have influenced the variation in diversity between sites, as Poole Bay is located to the west of the Isle of Wight you might expect higher species diversity compared to the east of the Isle of Wight due to the boundary zone for southern (Lusitanian) and northern (Boreal) species (Lewis, 1964). The increase species richness at Poole may also be due to differences in sea temperatures (Blight et al., 2009) and rock type may also have been influential. Softer rocks, such as limestone, naturally weather to create crevices and rough surfaces, whereas harder rock, such as granite, weather more slowly, leaving smooth, flat rock faces that are less favourable to species settlement and colonisation (Moschella et al., 2005). The quarrying process of cutting rock to size also produces smooth surfaces with little surface heterogeneity and so until significant weathering occurs, surface roughness will remain low (Coombes et al., 2011, 2015). The increased heterogeneity resulting from the treatments on the granite boulders at Runswick Bay enhanced the ability of organisms to colonise the granite boulders resulting in a marked increase in richness, abundance and diversity. Although variation in species richness has previously been observed on the inside and outside faces of limestone boulders used for rock groynes (Sherrard et al., 2016), this could not be assessed in this study due to inaccessibility of the inner faces. In addition the new species recorded in the holes and grooves may have been present in other un-sampled/ inaccessible areas on the rock armour.

Whilst a significant increase in number of barnacles occupying the grooves was observed on both shores (Table 6.3), this was not the case for the holes treatment. Barnacle settlement has been shown to be greater on rough surfaces (Hills et al., 1999; Berntsson et al., 2000, 2004; Menge et al., 2010), whilst mobile intertidal snails (e.g. *Littorina saxatilis*) actively select a groove or hole in a rock compared to a bare rock surface with no refuge (Pardo & Johnson, 2004). In the current trials, newly settled and mobile species favoured treatment areas over the bare rock faces. The treatments used in the current trial not only introduced additional substrate heterogeneity and rugosity, but also created areas of water retention (Figure 6.4). The lack of water retention and available refuges on artificial structures has previously been shown to result in reduced species richness (Bulleri & Chapman, 2004; Coombes et al., 2011; Firth et al., 2013b). In artificial rock pools created in a granite breakwater, Evans *et al.,* (2015), revealed that artificial pools supported equivalent species richness to the nearby natural rock pools, and the artificial pools were shown to create suitable habitat for species previously absent from the artificial structure at mid-shore height. The results here support this, as new species were also recorded in the holes and grooves at both sites that were previously absent from the boulders. Firth et al., (2013a) found that rock pools in artificial structures have a more pronounced effect on species richness in both the mid- and upper-shore zones. This suggests that modifications will have the greatest impact in the upper and mid shore habitats.

Limpets, however, did not show an increase in abundance with all treatments, which was attributed to the small amount of space in the holes, resulting in a limited size and abundance of individuals able to utilise them (See Methods section 6.2a for dimensions). At Poole, the number of limpets was significantly higher in areas which included the holes treatment, but the same effect was not observed at Runswick Bay. This could be explained by the lack for food available on the boulders. Furthermore, the grooves at Poole Bay regularly trapped stones, shells and sand which could both encourage and deter species from colonising (Liversage et al., 2017). The additional refuge created by shell and stone debris could facilitate development of algal spores (Bulleri, 2005a) and colonisation by small gastropod snails, yet could prevent refuge for large species such as limpets and fish. Overall, the use of these simple treatments had a positive effect on richness and diversity and enhanced the colonisation of common rocky shore species.

The reduced abundance of mobile fauna has previously been noted on artificial structures which results from low habitat heterogeneity and limited refugia (Chapman, 2003). Here, the addition of holes and grooves resulted in previously absent mobile fauna to be recorded on the groynes, including fish (*Lipophrys pholis*) and crabs (*Carcinus maenas*) in the holes of the Poole treatment. At Poole, the limpets (especially juveniles typically less than 16 mm) favoured the holes that acted as refugia until they had outgrown the hole, when they potentially migrated onto the surrounding rock surface. In the Azores, Martins et al., (2010) showed that holes can be used to successfully attract and harvest limpets for human consumption. Several algal species that attached to the rough textures within the grooves were absent on the bare rock faces. The creation of rough surfaces as a consequence of these interventions allowed algal spores to attach and ‘escape’ due to the refuge provided from predators, dislodgement and desiccation (Hawkins, 1981). The presence of macrophytes such as *Fucus* spp. will encourage subsequent mobile fauna, as the alga provides refuge from predators and desiccation (Christie et al., 2009).

The community establishment of an artificial structure will be dependent on the time of construction as priority effects are known to influence the arrival of propagule supply and subsequent community development (Hall, 2015). As coastal defence structures are commonly constructed in high wave energy environments, the communities formed on hard structures can be stripped back to a bare substratum during storm events (Sousa, 1979). Disturbance can be a key factor in controlling communities on artificial structures, especially in high energy environments. Structures which are built or suffer disturbance events in the spring or summer are likely to attract the largest variety of species due to greater larval supply (Minchinton & Scheibling, 1991; Jonsson et al., 2004). The development and survival of these communities will depend on the subsequent priority effects which are determined by biological and environmental conditions (Hall, 2015). Consequent changes in communities could be observed in subsequent months and years due to succession and disturbances, reinforcing the need for long term monitoring (Sheehan et al., 2013).

It has been established that artificial structures support less diverse communities than natural rocky shores (Chapman & Bulleri, 2003; Bulleri & Chapman, 2004; Moschella et al., 2005; Glasby et al., 2007; Vaselli et al., 2008; Firth et al., 2013b). Following an initial colonisation of microbial film, structures are colonised by larger opportunistic species such as *Ulva* spp. with subsequent community development then dependent on local conditions and propagule (larvae and spores) supply (Benedetti-Cecchi, 2000a). However, in the current study, the holes and grooves trials resulted in an increase in richness and diversity, irrespective of boulder type (See Table 6.2 and

Figure 6.3), indicating that even simple measures can have a beneficial effect on biodiversity. The nature of the enhancement technique also means that this can be implemented at any stage during the life history of the coastal defences, adding biodiversity to existing structures as well as being incorporated into new ones.

In contrast, there has been concern that artificial structures can increase the spread and abundance of non-native species (Bulleri & Airoldi, 2005), however in the current study the number of non-native species recorded at both sites was low. No non-native species were recorded at Runswick Bay (either in previous baseline surveys, treatments or controls). The barnacle *A. modestus* was found in both the holes and grooves treatments in Poole Bay but in numbers comparable to control areas. The increased interspecific competitive and predatory interactions resulting from higher species diversity associated with these treatments may limit populations of invasive species on these structures (Levine, 2000) however this was not confirmed at the scale of these experiments.

Species that are expanding their geographic range in response to rising temperatures (climate migrants such as *Gibbula umbilicalis* (Keith et al., 2011)) may benefit from such treatments. Both the warm-temperate barnacle species *Perforatus perforatus* and sea anemone *A. viridis* were found in the holes and grooves at Poole, but nowhere else on the groynes. The increased surface texture created by the treatments could facilitate further expansion of climate migrants as they provide refugia (Bourget et al., 1994) and could therefore promote establishment.

It is important to carefully consider the rationale for ecological enhancement of artificial structures prior to creation and installation. For example is the requirement as mitigation for habitat loss elsewhere in the region or are they primarily for an educational resource and local tourism? The interest shown by the general public illustrates that these techniques can add value to these schemes by improving biodiversity and visitor engagement and awareness (Morris et al., 2016).

Conclusions

This trial has successfully demonstrated that increasing habitat heterogeneity and providing water retention on granite and limestone rock armour can promote and encourage biodiversity on artificial structures. This study demonstrates how ecologists, coastal managers and engineers can work together to produce low cost interventions which produce successful results. These techniques can be used at any stage and are suitable for use in high wave energy environments where tiles or flowerpots would not be feasible. The granite interventions cost approx. £55/m^3 and the limestone interventions cost £10/m^3 however if these techniques were used during the initial construction phase then larger machinery could reduce the cost and time taken. In addition the correct positioning of quarried boulders can also create habitats to maximise water retention, for example where ‘blast lines’ or holes are already present. Future projects should upscale these smaller trials to large defence schemes, and aim to include a variety of sizes and depth of holes and grooves to further increase species richness and diversity of larger mobile species. Collaboration between ecologists and engineers is needed to develop multifunctional structures which can protect the land from coastal erosion and also create suitable habitat for marine organisms.

7 Shelving the Coast with Vertipools: Retrofitting Artificial Rock Pools on Coastal Structures as Mitigation for Coastal Squeeze

Abstract

Coastal squeeze caused by sea level rise threatens the size, type and quality of intertidal habitats. Along coastlines protected by hard defences, there is a risk that natural rocky shore habitat will be lost, with remaining assemblages characteristic of hard substrata confined to sea walls and breakwaters. These assemblages are likely to be less diverse and different to those found on natural shores as these structures lack features that provide moist refugia required by many organisms at low tide, such as pools and crevices. Yet engineering solutions can help mitigate the impacts of sea level rise by creating habitats that retain water on existing structures. A feasibility study retrofitted five concrete-cast artificial rock pools (‘Vertipools’) onto a vertical seawall on the south coast of England. After 3 years, the artificial pools increased the species diversity of the sea wall and attracted mobile fauna previously absent, including fish and crabs. The Vertipools had assemblages which differed significantly from natural rock pools at low water neaps and supported different functional groups including predators and grazers. Although disturbance of algal assemblages on the seawall from the retrofitting process was still evident after 3 years, succession to full canopy cover was underway. Collaboration between policy makers, ecologists, children and artists produced an ecologically sensitive design that delivered substantial benefits for biodiversity and that could be adapted and scaled-up to both mitigate habitat loss and enhance coastal recreational amenity.

7.1 Introduction

Coastal protection provided by seawalls forms a barrier between the land and the sea, preventing the natural migration of the coastline. These barriers, coupled with increased sea levels, are resulting in “coastal squeeze”, which occurs when the high water mark is fixed by a defence structure and the low water mark is moving landwards due to sea level rise, resulting in substantial losses of intertidal habitat (Pontee, 2011).

The construction of coastal defence structures results in the steepening of the shore profile (Jackson & McIlvenny, 2011) that creates less space for colonisation and a compressed species zonation (Kendall et al., 2004). This coupled with the lack of water retention which would naturally occur in crevices and pools (Firth et al., 2013b) results in a poor quality habitat. Habitat heterogeneity is also generally absent on most artificial structures; in contrast, natural rocky shores have a high variety of surface textures, crevices, overhangs and pools which provide suitable refugia and habitats for a diverse range of species (Connell, 1972; Underwood et al., 2008).

Water retention is important on a rocky shore as it creates refugia from desiccation stress and predation during periods of low tide (Firth et al., 2014a; White et al., 2014). Although the physico-chemical composition of rock pools is known to fluctuate diurnally and seasonally with changes in temperature, pH, salinity and oxygen saturation, these fluctuations are not as extreme as on the emergent rock surfaces (Daniel & Boyden, 1975; Metaxas & Scheibling, 1993). Rock pools are known to extend the limits of distribution for intertidal species, including larger brown algae (*Fucus* spp.), limpets and mussels (Green, 1971). Photosynthesis of algae within the pools can influence oxygen and carbon dioxide levels, which in turn affect the pH of the water (Metaxas & Scheibling, 1993; Björk et al., 2004). Intertidal fish also use rock pools as habitats, but these decline in abundance as shore height increases (Bennett & Griffiths, 1984; Zander et al., 1999). White et al, (2014) found that more complex rock pools with ledges and algal cover resulted in higher abundances and diversity of intertidal fish.

Ecological enhancement schemes that integrate ecology with engineering, can create potential solutions to mitigate for low habitat heterogeneity through producing multifunctional structures that provide coastal protection which provide suitable habitats for marine species (Dafforn et al., 2015; Firth et al., 2016b; Morris et al., 2016; Evans et al., 2017). Trials have indicated that creating artificial structures which increase water retention and habitat heterogeneity can create opportunities for colonisation by a variety of species (Chapman & Blockley, 2009; Browne & Chapman, 2011; Firth et al., 2014a; Evans et al., 2015). Browne & Chapman (2011, 2014) deployed precast flowerpots at different tidal heights on a seawall in Sydney to mimic natural rock pools, although some flowerpots were dislodged through wave action, the remaining ones increased the biodiversity of the seawall by attracting novel sessile species. Morris *et al* (2017) investigated the larger scale effects of the flowerpots by studying the mobile communities associated with the flowerpots. In addition, they added artificial turf to some of the pots in order to see the effects on the native and non-native sessile communities. The study found higher densities of mobile and sessile species in pots without artificial turf, implying that the turf prevented particular species from colonising the pots. The outcomes of experimental trials are strongly affected by local conditions and motivations (Airoldi et al., 2005), therefore development of new techniques and solutions are important in order to meet the needs of local communities and developers.

As part of the community science project '*Shelving the Coast*' on the Isle of Wight, southern England, artists, school children and ecologists designed and created a series of structures subsequently named and referred to here as 'Vertipools'. These pre-cast concrete artificial rock pools aimed to vertically extend the intertidal zone to mitigate the effects of sea level rise, which could result in the loss of the existing intertidal rock pools. The Vertipools were designed to be attached to coastal structures, such as vertical seawalls and groynes, and are V-shaped (to deflect wave energy) with hollow insides (to retain water at low tide). The exterior patterns were designed by local primary school aged 5 to 7 as part of an educational project on coastal squeeze. The educational programme involved three activities firstly an animation project which enabled the children to understand the impacts of sea level rise and the concept of coastal squeeze. Followed by song writing and recording exercise based on the topic of coastal squeeze and finally the designing of the “Vertipool” exterior surfaces. The children all created an exterior design for the Vertipools using the knowledge they had gained from the education activities.

To determine whether the establishment of Vertipools could create similar habitats to nearby natural rock pools on the shore and minimise disturbance to existing habitats on the sea wall the following hypotheses were tested:

1) Vertipools will support similar species richness and functional groups as natural shore pools after 3 years;
2) The pH, temperature and salinity of the water in the Vertipools will be similar to the natural pools.
3) Vertipools will support greater species richness than the seawalls after 3 years.

In order to assess the impact of Vertipool construction on existing assemblages on the seawall:

4) The disturbed area of the seawall will have similar species richness and total abundance to the undisturbed seawall after 3 years.

7.2 Methods

7.2.1 Study Site

The site at Bouldnor is located on the south coast of England approximately 5 km east of Yarmouth on the north west coast of the Isle of Wight (50°42'27.5"N 1°28'57.1"W). The shore is moderately sheltered with a north facing aspect and a mean tidal range of 2m. This stretch of coast has been heavily modified including the vertical concrete seawall, constructed in 1985, which had well-established zones of marine algae dominated by the brown seaweeds *Ascophyllum nodosum* and *Fucus spiralis*. Below the sea wall, the shore is truncated, consisting of limestone boulders and a few natural rock pools surrounded by mobile mixed sediments. This site was chosen due to the low risk of public interference and ease of access for installation and monitoring.

7.2.2 Vertipool description

During September 2013, five concrete wooden-cast Vertipools were installed between Mean Tide Level (MTL) and High Water Neaps (HWN) on the vertical concrete seawall. The outside of each Vertipool was hand sculptured using wet cement to incorporate the design of the school children (Figure 7.1a). The Vertipools weigh 50 to 70 kg and are 900 mm at their widest, 610 mm in height, protrude a maximum 400 mm from the seawall and have an undulating pool depth of 10-150 mm. Three 20 mm diameter M20 stainless steel coach bars attached to a T-shape steel plate were cast into the back of the Vertipools for attachment to the seawall. Three holes were made in the seawall using a 25 mm diameter SDS drill bit to a depth of 150 mm and filled with a marine grade resin bonding agent (Fischer Resin Mortar), before attaching the Vertipools.

Figure 7.1 Images of the Vertipools; a) Vertipool and cleared area immediately after installation in 2013, showing detail of exterior design, b) Vertipool after 2 years, c) and d) positioning of Vertipools on seawall, e) and f) Vertipools after 3 years.

7.2.3 Assemblage Monitoring

After 3 years, all fauna and flora were sampled from each habitat using in-situ non-destructive sampling techniques and organisms were identified to the lowest taxonomic resolution possible.

Vertipools

Percentage cover of flora and counts of sessile and mobile fauna were recorded on the inside and outside of each Vertipool separately. The data collected from the assemblages on the outside of the Vertipools were compared to the assemblages recorded on the seawall (n=5) and the assemblages found inside the Vertipools were compared to the natural shore pools (n=5).

Natural Shore Pools

To compare the assemblages in the inside of the Vertipools with the pools on the natural rocky shore, five pools of similar surface area and water depth were located on the shore. As no natural pools were present on the high or mid shore the natural pools were situated at Mean Low Water Neap tide mark (MLWN). Percentage cover of flora and counts of sessile and mobile fauna were recorded on the inside of each natural pools (N=5).

A meter (HACH, Manchester, UK) and refractometer (Atago ATC-S/Mill-E) were used to measure water temperature, pH and salinity of the water inside the Vertipools and in the water inside the natural shore pools.

Control Seawall

Ten 20 x 20cm quadrats were used to record the percentage cover of algae and counts of sessile and mobile fauna on the undisturbed vertical seawall (>2m distant) adjacent to the Vertipools. This data was compared with the data collected on the disturbed sea wall affected by the retrofitting process.

Disturbed Seawall

To measure the effect of retrofitting on the existing algal assemblage on the vertical sea wall, areas cleared at the time of installation (referred to as “Disturbed Seawall”, Figure 7.1a) were subsequently monitored using two 20 x 20cm quadrats placed either side of each of the five Vertipools to record percentage cover of flora and counts of sessile and mobile fauna (N=10).

7.2.4 Statistical analyses

All data was tested for normality and equal variances and all t-tests were run using R v3.2.4 (R Core Team, 2016). Hypothesis 1, that Vertipools will support similar assemblages and functional groups as natural shore pools was tested using data collected after three years. Firstly a t-test was used to test the differences in species richness between Vertipools and shore pools. PRIMER-E was used to create a Bray Curtis similarity matrix on square-root transformed data. Assemblages were presented visually using Multidimensional Scaling (MDS) and ANOSIM was used to test for differences in assemblages between habitats. A SIMPER analysis was then applied to identify which species accounted for most of the variation in assemblages between habitats. In order to investigate the variation in functional groups, species were classified into morph-functional groups: Canopy algae, Sub-canopy algae, Turf algae, Filter feeders, Grazers and Predators (Arenas et al., 2006; Firth et al., 2014a). The functional groups were used to create a Bray Curtis similarity matrix on square-root transformed data. ANOSIM and SIMPER were used to highlight the variation in functional groups between Vertipools and shore pools. An MDS with a Pearson’s correlation overlay was used to illustrate the correlations between functional groups and habitats. In addition the functional groups were used to highlight the seasonal variation of communities within the Vertipools during spring 2015-winter 2016.

The second hypothesis that the pH, temperature and salinity of the water in the Vertipools will be similar to the natural pools was tested using a two sample t-test with habitat as the factor.

To test hypothesis 3 that Vertipools will support greater species richness than the seawall after 3 years, the data recorded from the inside and outside habitats on each Vertipool were combined and averaged (N=5). A two sample t-test was used to test the differences in mean species richness between habitats (Vertipool and control seawall). Assemblages were explored using PRIMER in order to create an MDS plot and run an ANOSIM and SIMPER analysis so as to determine the species contributing to the variation in community structure.

In order to test hypothesis 4 that the disturbed area of the seawall will have similar species richness and total abundance to the undisturbed seawall after 3 years, a two sample t-test was used to test the differences in mean species richness and total abundance between habitats (Disturbed seawall and Control seawall). In order to visually asses the similarity of communities, an MDS plot was used in combination with an ANOSIM and SIMPER to highlight the variation in species abundance.

7.3 Results

After 3 years, 16 species were recorded in the Vertipools, 8 species on both the control and disturbed seawall and 11 species in the natural shore pools. During the study, the algal cover on the exterior of the Vertipools developed from a community dominated by opportunistic green algae (*Ulva* spp.) into a dense over-hanging fucoid canopy. The assemblages inside the Vertipools comprised of fucoids, filamentous and branching algae. Mobile species observed inside the Vertipools included fish (*Lipophrys pholis*), shore crabs (*Carcinus maenas*), and gastropods *Patella vulgata* and *Littorina obtusata*, all of which were absent in the natural shore pools and on the seawall (with the exception of *L. obtusata*). The shore crab *C. maenas* was observed inside the

Vertipools at various life stages including juvenile, adult and freshly moulted; an old exoskeleton was found inside the pool indicating recent ecdysis.

7.3.1 Vertipools vs. Natural shore pools

A comparison of assemblages within the Vertipools and shore pools after 3 years revealed that the species richness was significantly greater in the Vertipools (Table 7.1; Figure 7.2). The community similarity was also significantly different (ANOSIM, Global R= 0.644, P=0.008), as illustrated in Figure 7.3 the shore pools showed more variation in community composition in comparison to the Vertipools. The SIMPER routine identified eighteen species which accounted for 99.9% of the overall 90.87% dissimilarity between samples (Table 7.2). The most abundant sessile species in the Vertipools were *Fucus spiralis, Ectocarpus* sp, *Chaetomorpha* sp., *Ulva lactuca,* and *Spirorbis spirorbis* whereas *Polysiphonia* sp., *Halopithys incurva, Chondrus crispus* and *Ceramium* spp. were most abundant in the shore pools (Table 7.2). Mobile fauna were only recorded in the Vertipools and these were *Carcinus maenas, Littorina littorea* and *Littorina obtusata.*

Table 7.1 T-test for difference in mean species richness (S) between habitats- Vertipools and Shore pools after three years (*** = P<0.001, ** = P<0.01,* = 0.05 NS= Not significant).

Source	***df***	**t**	**p**	**Significance**
Habitat	8	3.73	0.005	**

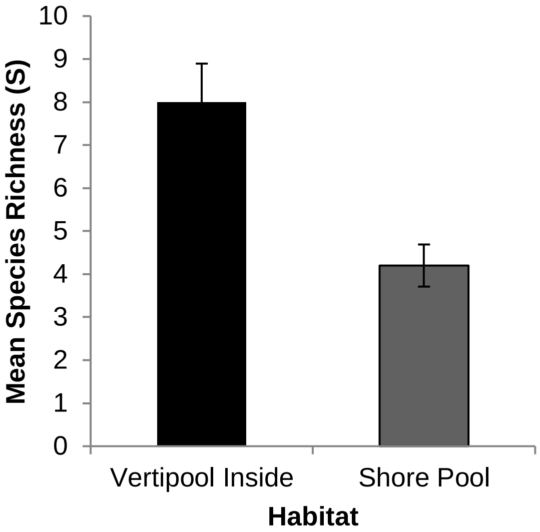

Figure 7.2 Mean species richness recorded inside the Vertipools compared with the shore pools after three years (+/- S.E).

Table 7.2 Differences in mean abundance of species recorded in the shore pools and Vertipools (percentage cover (%) or counts (c)) after three years. % contribution to the dissimilarities between assemblages in each habitat (SIMPER, Average dissimilarity= 90.87%)

Species	**Shore Pools**	**Vertipools**	**% Contribution**	**Dissimilarity/ s.d.**
Fucus spiralis (%)	0	54.33	30.64	1.88
Polysiphonia sp *(%)*	28.60	1.60	15.94	1.25
Ectocarpus sp *(%)*	4.00	22.33	12.12	1.24
Chaetomorpha sp*. (%)*	0	9.33	6.43	0.68
Spirorbis spirorbis (c)	0	7.73	5.21	0.74
Halopithys incurva (%)	8.60	0	4.66	0.49
Chondrus crispus (%)	8.00	0	4.48	1.09
Ceramium sp*. (%)*	6.80	3.07	4.40	1.18
Ulva linza (%)	3.40	4.2	3.56	1.19
Ulva lactuca (%)	4.00	5.13	3.04	1.23
Fucus serratus (%)	2.00	4.67	2.97	0.81
Sargassum muticum (%)	4.00	0	2.42	0.49
Mastocarpus stellatus (%)	3.40	0	1.99	0.70
Cladophora rupestris (%)	2.00	0.80	1.68	0.62
Carcinus maenas (c)	0	0.40	0.25	0.82
Ascophyllum nodosum (%)	0	0.20	0.10	0.49
Littorina littorea (c)	0	0.07	0.04	0.49
Littorina obtusata (c)	0	0.07	0.04	0.49
Idotea granulosa (c)	0	0.07	0.04	0.49

Functional Groups

The functional groups of species within shore pools and the Vertipools were also significantly different (ANOSIM, Global R = 0.372, P = 0.016). The SIMPER revealed the three functional groups which contributed 99% towards the overall 61.79% dissimilarity between habitats- Canopy algae, Sub canopy algae and Filter feeders. Overall there was a higher abundance of all functional groups in the Vertipools, include predators and grazers which were only found in the Vertipools (SIMPER, Average dissimilarity= 46.44%). The MDS illustrates the dominance of Grazers, Predators, and Filter feeders in the Vertipools compared to the natural pools (Figure 7.3).

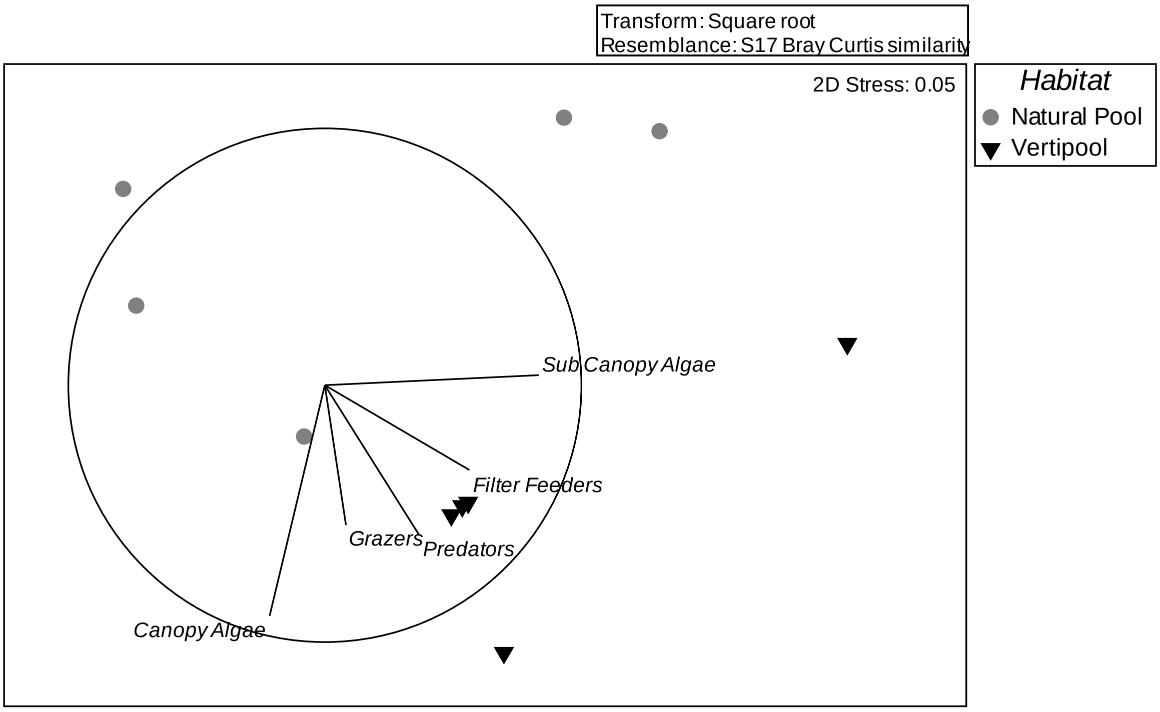

Figure 7.3 Multidimensional Scaling plot of species assemblages with the functional groups overlaid as a Pearson's correlation vector (r <0.3).

The seasonal variation within the Vertipools highlights the succession from a Sub Canopy algae dominated community to a Canopy algae dominated community (Figure 7.4) alongside the arrival of filter feeders in March 2016.

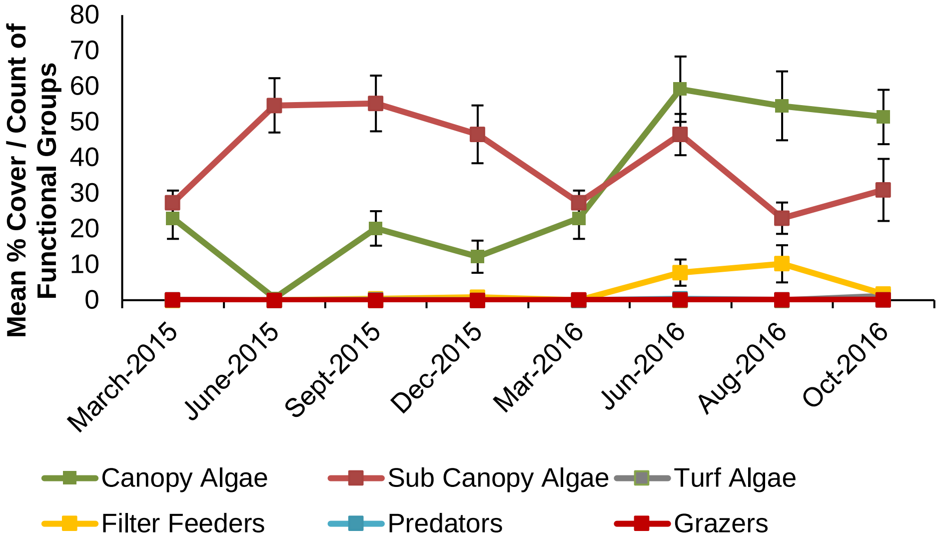

Figure 7.4 Seasonal variation in mean % cover/ count of functional groups inside the Vertipools between March 2015 and October 2016 (+/- S.E.)

Abiotic conditions

The abiotic measurements taken from the shore pools and Vertipools showed the Vertipools were generally cooler than the shore pools. There was a significant difference in temperature between habitats, yet the pH did not differ significantly between habitats (Table 7.3, Figure 7.5). The salinity of all shore pools were 36 ppt and the salinity of all Vertipools were 35 ppt (Figure 7.5).

Table 7.3 Two sample t-test results for the comparison of pH and temperature between the shore pools and the Vertipools after 3 years (NS= Not significant, ***= P<0.001).

	pH				Temperature			
	df	t	p	Sig	df	t	p	Sig
Habitat	8	1.19	0.26	NS	8	8.47	<0.001	***

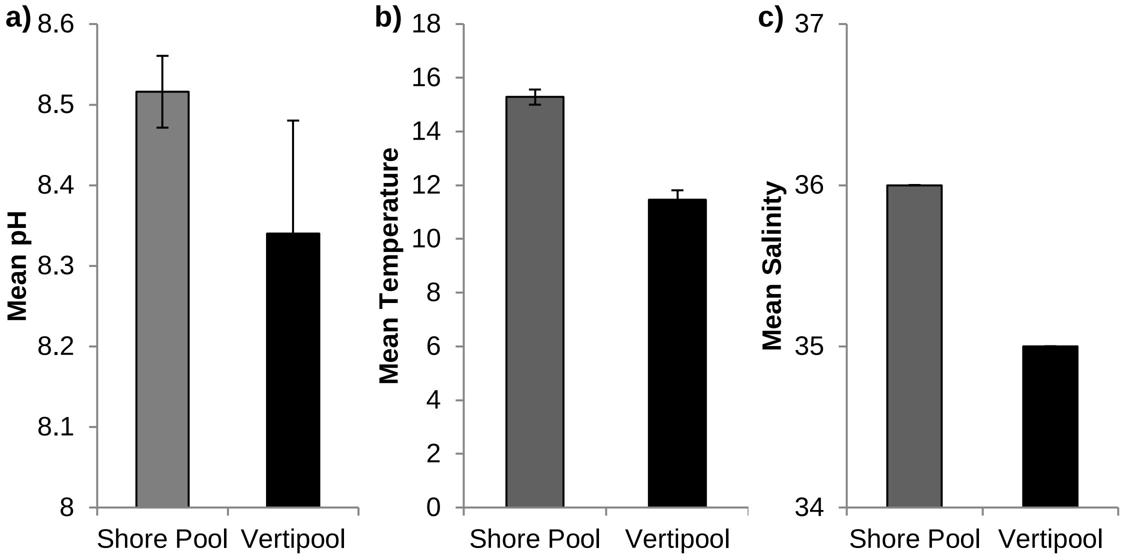

Figure 7.5 Mean a) pH b) temperature and c) salinity for the shore pools and Vertipools after 3 years (N=5, +/- S.E.)

7.3.2 Vertipool vs. Seawall

After three years the combined data from both inside and outside of the Vertipool had significantly higher species richness than the adjacent seawall (Table 7.4, Figure 7.6). Eight species were found to be unique to the Vertipools; these were predominantly mobile species (Table 7.5).

Table 7.4 T-test for difference in mean species richness between habitats- seawall and Vertipool (***=P<0.001, **=P<0.01,*=0.05 NS= Not significant).

Source	df	t	p	Significance
Habitat	8	-4.35	0.002	**

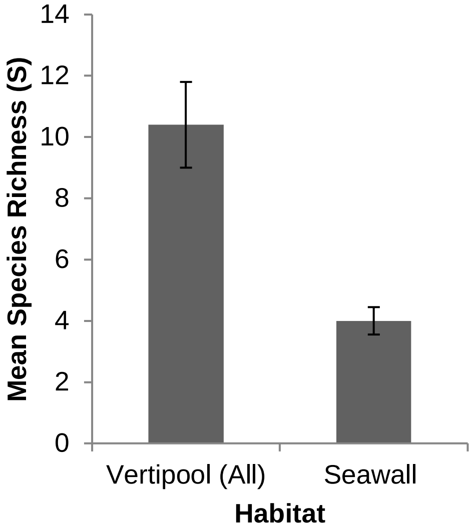

Figure 7.6 Mean species richness after three years recorded in each habitat- Vertipool (Inside and Outside combined) and Seawall (Control) (+/- S.E.).

There was also a significant difference in assemblage composition between the Vertipools and the seawall (Figure 7.3a; ANOSIM, Global R = 1.00, P = 0.008). The SIMPER analysis revealed that 78% of the overall 86.31% dissimilarity between habitats was due to four algal species; *A. nodosum, Rhodothamniella floridula, Fucus spiralis* and *Cladophora rupestris. A. nodosum, R. floridula* were most abundant on the seawall, whereas *F. spiralis* and *C. rupestris* were most abundant on the Vertipool (Table 7.5).

Table 7.5 Differences in mean abundance of species recorded in the Vertipools to the Seawall (percentage cover (%) or counts (c)) after three years. % contribution to the dissimilarities between assemblages in each habitat (SIMPER, Average dissimilarity= 86.31%).

Species	**Vertipool**	**Seawall**	**% Contribution**	**Dissimilarity/ s.d.**
Ascophyllum nodosum (%)	0	90.00	34.14	7.48
Rhodothamniella floridula (%)	3.00	67.00	24.32	5.93
Fucus spiralis (%)	54.83	6.00	18.29	2.83
Cladophora rupestris (%)	11.67	42.00	11.85	2.02
Ulva linza (%)	16.83	0	6.32	2.81
Ulva lactuca (%)	3.07	0	1.17	1.16
Chaetomorpha sp. *(%)*	2.83	0	1.11	0.78
Fucus serratus (%)	2.60	1.00	1.04	0.94
Catenella sp. *(%)*	2.10	1.00	0.89	0.98
Spirorbis spirorbis (c)	0.90	0	0.34	1.00
Semibalanus balanoides (c)	0.60	0	0.23	1.00
Idotea granulosa (c)	0.30	0	0.11	0.96
Polysiphonia sp *(%)*	0.13	0	0.05	0.69
Patella vulgata (c)	0.10	0	0.04	0.73
Sargassum muticum (%)	0.10	0	0.04	1.20
Carcinus maenas (c)	0.07	0	0.03	0.80
Austrominius modestus (c)	0.07	0	0.02	0.49
Littorina obtusata (c)	0.03	0	0.01	0.49

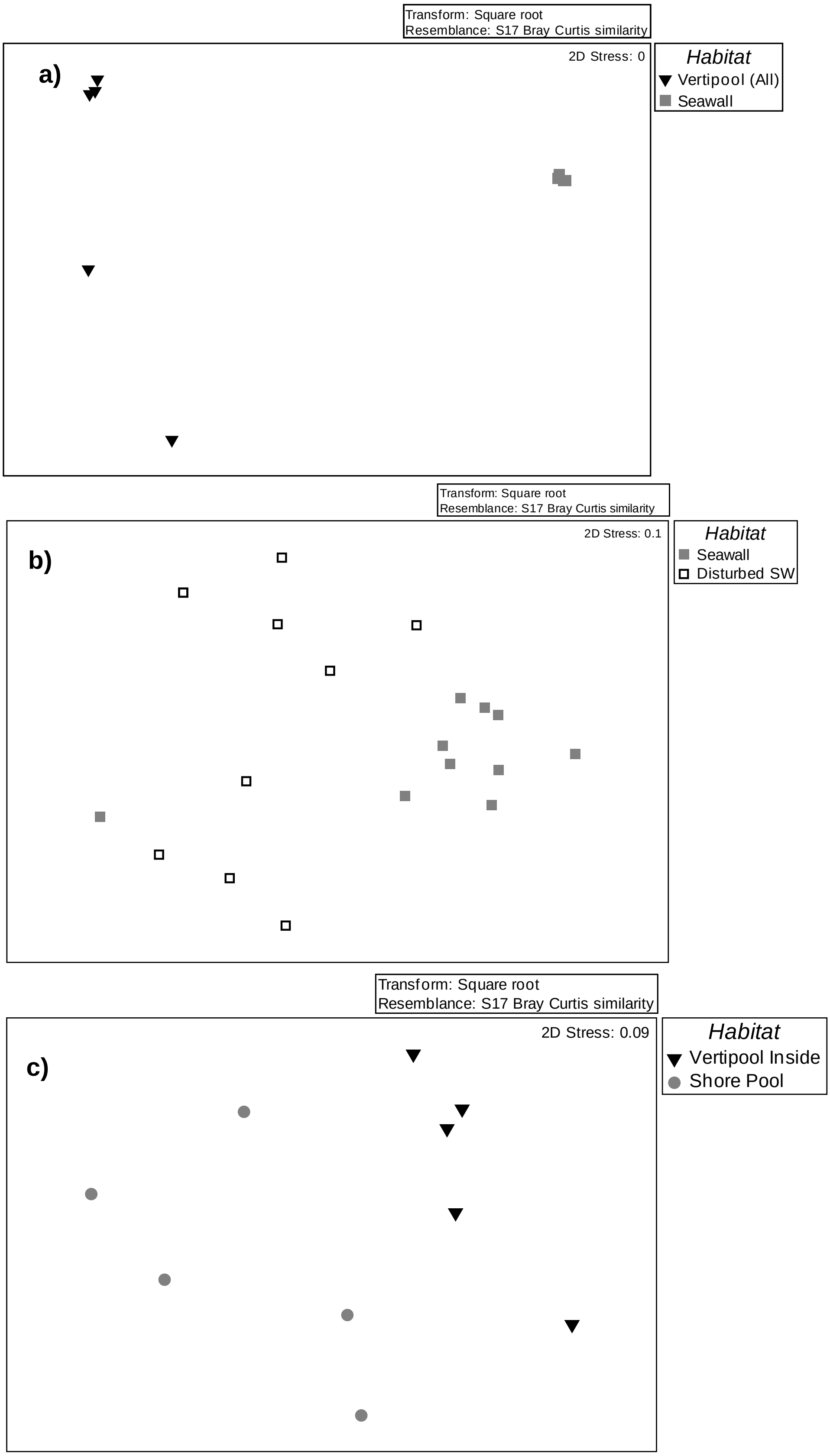

Figure 7.7 Multidimensional Scaling Plot (MDS) comparing assemblages within habitats a) Vertipools (Inside and Outside) and Control Seawall, b) Control seawall and Disturbed Seawall c) Vertipool (Inside) and Shore Pool

7.3.3 Disturbed seawall vs. undisturbed seawall

After three years, the disturbed seawall areas had recovered to similar species richness to that of the undisturbed seawall, although the total abundance of biota on the disturbed seawall was significantly lower than on the undisturbed seawall (Table 7.6, Figure 7.8).

Table 7.6 T-test for difference in mean species richness (S) and total abundance (N) between habitats- control seawall and disturbed seawall (*** = P<0.001, ** = P<0.01,* = 0.05, NS= Not significant).

	Species Richness (S)				Total Abundance (N)			
Source	df	t	p	Significance	df	t	p	Significance
Habitat	18	1.51	0.14	NS	18	6.92	<0.001	***

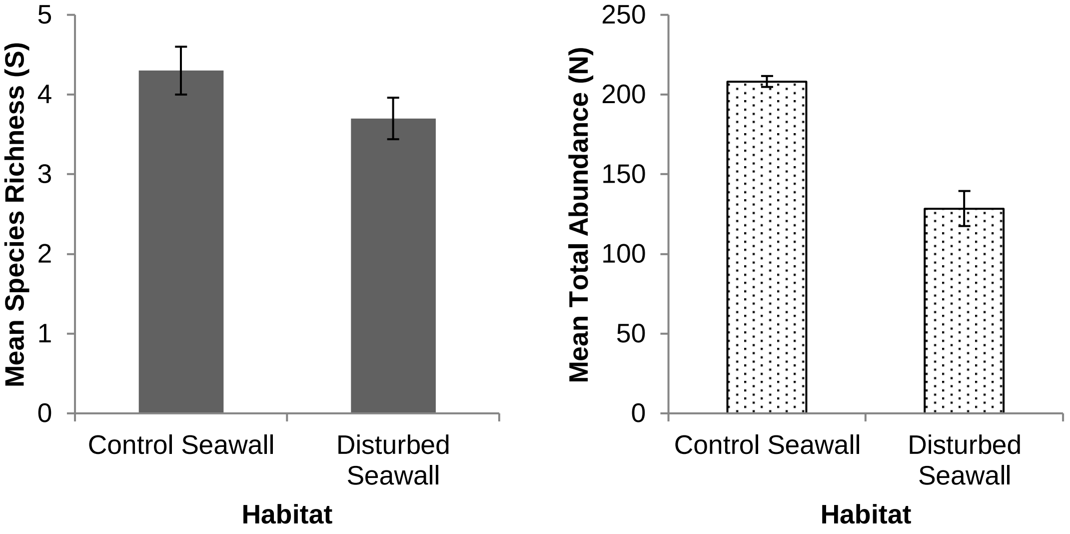

Figure 7.8 Mean a) species richness and b) total abundance of species recorded on the disturbed seawall and the control seawall after 3 years (+/- S.E).

In addition, there was also a significant difference in assemblage composition between the disturbed and undisturbed seawall (ANOSIM, Global R = 0.536, P = 0.001). The MDS plot illustrates the dissimilarity between the assemblages of disturbed and undisturbed areas of the seawall (Figure 7.7b). The SIMPER routine revealed that *A. nodosum, R. floridula* were most abundant on the undisturbed seawall whereas *F. spiralis, C. rupestris* and *F. serratus* were more abundant on the disturbed seawall, contributing to 90% of the overall 45.96% dissimilarity between the habitats (Table 7.7)

Table 7.7 SIMPER analysis on disturbed vs. control seawall, mean abundance (%) of species contributing to the 90 % dissimilarity between habitats (Cum %= cumulative percentage).

Species	Control Seawall	Disturbed Seawall	Contribution %	Diss / SD	Cum %
Ascophyllum nodosum	85.50	2.60	39.13	3.32	39.13
Rhodothamniella floridula	60.50	26.50	16.39	1.84	55.52
Fucus spiralis	12.00	19.00	14.69	1.16	70.22
Cladophora rupestris	45.00	69.00	11.22	1.41	81.44
Fucus serratus	2.50	7.00	8.60	0.90	90.04

7.4 Discussion

The water retention and increased surface texture provided by the Vertipools have created habitat which is absent from the existing sea wall, enabling rock pool species to survive on the seawall. The Vertipools increased the species richness and altered the composition of species on the seawall supporting communities more characteristic of natural rocky shores (Metaxas & Scheibling, 1993; White et al., 2014). Yet, assemblages in the Vertipools differed to the natural pools due to the earlier stage of colonisation, higher tidal level and variation in sedimentation. The angular design of the Vertipools allowed for waves to deflect off them which resulted in all of the Vertipools remaining attached to the seawall with no visible signs of damage. Unlike in previous studies where destruction of enhancement devices by wave action has been a problem

(Browne & Chapman, 2014). As sea levels rise and coastal squeeze becomes more severe (Pontee, 2011), limiting the refugia provided by natural habitat (Jackson & McIlvenny, 2011), it is probable that the Vertipools will become increasingly accessible to species currently surviving in natural pools at lower tidal levels. Therefore longer term changes in community structure in the Vertipools are expected.

Initially, the Vertipools were colonised by opportunistic green algae (*Ulva* Spp.), followed by a shift to fucoid algae on the exterior and branching/filamentous algae on the interior. This follows typical succession on a rocky shore (Benedetti-Cecchi & Cinelli, 1996; Benedetti-Cecchi, 2000b; Martins et al., 2007; Viejo et al., 2008), although it is hard to say whether or not the assemblages in the Vertipools have stabilised yet and a longer monitoring period is required (Browne & Chapman, 2014). Seasonal variation in assemblages has been observed in the Vertipools, with red filamentous algae appearing in the summer months (Christie et al., 2009) and barnacles recruiting in the spring (Jenkins et al., 2000). The close proximity to natural habitat and propagule supply may have facilitated colonisation at this site and locations with less spatial connectivity may take longer to colonise (Cowen & Sponaugle, 2009).

In comparison to natural rock pools, the Vertipools supported similar groups of species including fucoids, filamentous red and green algae in addition to mobile fauna. Crabs at various life stages have been recorded inside the Vertipools, including freshly moulted, indicating that the Vertipools are acting as natural rock pools by creating refuges for mobile fauna at low tide (Zander et al., 1999). Filter feeders and grazers were also found inhabiting the Vertipools; while grazers such as littorinids are commonly found in upper shore pools, filter feeders are predominantly found in low shore pools due to the need for regular exchange of water (Huggett & Griffiths, 1986). The variation in functional groups between natural and artificial pools is to be expected due to the different tidal heights, however the findings show that the Vertipools do have the potential to act as mitigation for sea level rise.

The Vertipool located at the greatest height on the seawall took longest to colonise, with the interior community consisting predominately of opportunistic algae (*Ulva* spp.), whereas the exterior was colonised by *F. spiralis*. Previously, the reduced number of organisms in high shore ecological enhancements has been linked to low recruitment levels (Browne & Chapman, 2014). Although, Firth et al (2016b) discovered that mean species richness was the same in upper and lower shore artificial pools created on a Shepherd Hill energy dissipation (SHED) unit in Ireland and the differences were found in the community composition of species. Destructive sampling was undertaken after 24 months of installation and they found that suspension feeders, grazers and carnivores were lacking in the upper pools, yet filamentous and foliose algae was abundant. The current study did not incorporate destructive sampling as a main focus of the project was community engagement and it was therefore not deemed appropriate.

Studies have shown that rock pools support a more diverse community than adjacent rock faces (Firth et al., 2013b, 2014a) on both natural shores (Firth et al., 2014a) and artificial structures (Chapman & Blockley, 2009; Browne & Chapman, 2014; Evans et al., 2015). As with natural rock pools found on the upper shore, the distributional limits of intertidal species extended higher up the shore due to the installation of the Vertipools (Metaxas & Scheibling, 1993). Mobile fauna such as crabs (*C. maenas)* and periwinkles (*L. obtusata),* previously absent from the seawall, were found inside the Vertipools on multiple occasions. It is possible that the temperatures of the Vertipools were cooler than the natural pools due to the shading created by the seawall, limiting the amount of direct sunlight (Blockley & Chapman, 2006). Water temperatures are likely to vary by location due to aspect and exposure and the seawall used in this study was north facing which resulting in prolonged periods of shading. A south facing seawall may experience harsher conditions resulting in higher desiccation effects. Shore height, pool volume, surface area, depth, shading and drainage are known to impact the physico-chemical composition of rock pools (Daniel & Boyden, 1975; Metaxas & Scheibling, 1993; White et al., 2014). Whilst oxygen levels were not recorded, the presence of fish, crabs and other fauna inside the Vertipools indicated suitable oxygen levels, despite the potential for oxygen sags at night due to seaweed respiration.

In the present study, the shoreline and seawall were dominated by *A. nodosum* which is slow to recover after disturbance events (Jenkins et al., 2004), due to poor growth and recruitment mortality (Stengel & Dring, 1997). This study monitored how the retrofitting process affected the existing algal assemblages on the seawall. After three years the disturbed areas were recolonised by *F. spiralis* (19.00% cover) and *A. nodosum* (2.60% cover), with an understory of *R. floridula* (26.50% cover) and *C. rupestris* sp. (69% cover). However, as also shown by Jenkins et al. (2004), *A. nodosum* took longer to grow than *F. spiralis*. Overall, the early recolonisation of algae indicates that the retrofitting process is unlikely to have any long term impact on these assemblages.

As this study was a small scale trial project, the five Vertipools were only installed at one site, resulting in low spatial replication. Future studies will need to include trials at multiple locations with increased replication in order to determine wider scale benefits and impacts. These locations should also include both sheltered and exposed locations. One criticism of retrofitted objects is that they might reduce the structural integrity of the seawall (French, 2001). To date, however, no signs of damage or weakening have been recorded, although this will continue to be monitored over time. Moreover, cross disciplinary work needs to be conducted between engineers and ecologists to create multifunctional structures for the future (Dafforn et al., 2015; Firth et al., 2016a). Evidence has suggested that if ecological enhancement devices are aesthetically pleasing, the general public and coastal managers are more supportive of their use (Morris et al., 2016). Incorporating education and public engagement into habitat creation schemes is an excellent way to connect and educate the general public and school children on important issues such as coastal squeeze and sea level rise. The use of art to activate and engage the students on a complex topic worked extremely well and is recommended for future projects

Conclusions

Extending the intertidal zone vertically by creating suitable artificial rock pools for marine life to inhabit has been successful. Pools could potentially be installed and retrofitted on a variety of different coastal structures. If replicated more widely, these features have potential to mitigate for the impact of coastal squeeze and other physical disturbances that limit the size of the intertidal zone, such as coastal development. Combined with other interventions, such as the creation of holes and grooves to create refugia at different scales, habitat heterogeneity on these structures will increase species and functional diversity.

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8 General Discussion

8.1 Thesis overview and summary

The overall aim of the thesis was to quantify the variation in biodiversity on different artificial coastal structures compared to natural habitats and assess the magnitude of species interactions, connectivity and the potential for ecological enhancement. This aim was addressed through intertidal and subtidal surveys in order to quantify and monitor the ecological processes occurring on and around artificial structures, coupled with bio-physical models to determine the potential connectivity of marine biodiversity within the region.

8.2 Ecology of artificial coastal structures

The communities colonising many coastal artificial structures have been previously studied in detail (Connell, 2001; Thompson et al., 2002; Bacchiocchi & Airoldi, 2003; Bulleri & Chapman, 2004; Bulleri, 2005b), yet knowledge gaps are still apparent. Wooden groynes have been used historically for hundreds of years, yet to date no in depth research has been conducted into the communities colonising the structures. Although natural shores had a higher species diversity compared to the groynes, supporting general predications of others (Moschella et al., 2005; Pister, 2009; Firth et al., 2013b; Aguilera et al., 2014), contrary to hypothesis, this research discovered that the wooden groynes supported a greater number and diversity of species in comparison to the rock groynes and there were particular species preferences on both substrates. *Patella vulgata* and *Melaraphe neritoides* were most abundant on the rock groynes, whereas *Semibalanus balanoides, Austrominius modestus* and *Mytilus edulis* were most abundant on wooden groynes. In addition, there was a greater abundance of algae on the wooden groynes, particularly *Fucus spiralis* and *Ulva* spp. which will facilitate the survival of future colonisers through the provision of food and/or shelter.

Artificial structures may be of high importance in facilitating range extensions at local and broad geographical scales. In contrast to existing literature that has shown strong associations between artificial structures and non-native species (Bulleri & Airoldi, 2005; Vaselli et al., 2008; Dafforn et al., 2012), only three non-native species were recorded on the wooden and rock groynes, the barnacle *A. modestus,* the Pacific oyster *Crassostrea gigas* and the green alga *Codium fragilie,* all of which were found in comparable numbers on the nearby natural rocky shores. Several climate migrants were also recorded on both the natural shores and groynes, including *Perforatus perforatus, Patella depressa* and *Gibbula umbilicalis* all of which are known to be extending their range eastwards as a result of climate change (Herbert et al., 2003; Mieszkowska et al., 2005). Understanding the material preference for species could enable us to prevent the future spread of non-native species and allow us to conserve and encourage native species to colonise artificial structures. In urban environments, walls built from stone are typically colonised by rock outcrop species whereas walls constructed from steel or glass deter colonisation by typical wall flora (Lundholm & Richardson, 2010). The reason that opportunistic species such as non-natives colonise novel substrates is because they are better adapted to living in novel conditions due to their plasticity (Hill et al., 2002).

Evidence of significant regional variation in communities on artificial structures was also found during this study, which is unsurprising due to the variation in age, substrate type, length, height and local conditions. Chapter 2 also documents evidence that the tidal level to which an artificial structure extends was found to be an important factor in structuring the community and mid shore areas within the outer sections on groynes supported significantly more species than the upper shore areas. Clear zonation could be observed on the wooden groynes (Chapter 2, Figure 21b) with barnacles and limpets observed on the upper tidal heights and mussels observed on the lower tidal heights. This highlights the importance of tidal level in shaping the communities present on artificial structures as already demonstrated on natural rocky shore (Lubchenco, 1980; Raffaelli & Hawkins, 1996). Yet these structures can create novel distribution patterns, for example the provision of tall, hard substrate at the outer ends of the groynes allows species such as *M. neritoides* to colonise which would otherwise be absent to survive at further distances down the shore. Given time and adequate larval supply it appears that artificial structures which extend into Extreme Low Water Springs (ELWS) have the potential to support stable communities similar to those found on natural shores, yet there is still a lack of mobile predators recorded on the structures. If increased numbers of predators were to arrive they could have a large effect on altering the communities (Connell, 1961a, 1961b) and future experiments should be conducted to investigate the impacts of predators on artificial structure communities. High level of regional variation, both spatial and temporal makes it difficult to predict colonisation and potential impact on metapopulations

Chapter 3 documents that the settlement and recruitment processes involved in structuring the communities on wooden and rock groynes were comparable to those observed on natural rocky shores. This is vital to understand as when constructed in soft sediment environments, artificial structures become the only hard substrate available to colonise, which could alter the connectivity dynamics of the region (Bishop et al., 2017). Yet, this research found large regional variation between substrate preferences for settlement and recruitment of *S. balanoides* which reinforced the importance of site-specific and species-specific research (Burrows et al., 2010). As mentioned in Chapter 1 Section 1.2.2 various factors influence the colonisation of artificial structures, yet one factor which needs more investigation is the effect of maintenance. The age and maintenance of an artificial structures will impact on community establishment and stability of populations (Pinn et al., 2005). The wooden groynes within Region 2 were poorly maintained and very brittle resulting in lower recruitment of *S. balanoides* due to inadequate habitat provision (Herbert & Hawkins, 2006). If the loss of intertidal habitat continues as predicted we need to ensure artificial structures can provide suitable habitat in order to sustain reproductive populations with the purpose of maintaining the metapopulations of the region.

Connectivity

The association of mobile fauna such as fish and crustaceans with artificial structures had rarely been investigated. Through the use of Baited Remote Underwater Video (BRUV) and Stable Isotope Analysis (SIA), Chapter 4 documented a significant association of mobile fauna to piers. The provision of hard substrate created both a site for colonisation of epibiota which forms a food resource and also the provision of shelter from both predation and hydrodynamic conditions. There is evidence that the piers are acting as a corridor, linking the subtidal habitat with the intertidal areas, in which suitable foraging areas can be exploited. Wildlife corridors are an established concept in terrestrial ecology yet little effort has been made to apply the knowledge to marine environments (Krost et al., 2017). Hedgerows have been used in lowland agricultural landscapes to provide both habitat for resident species and resources for migratory species such as birds and mammals (Hinsley & Bellamy, 2000). The same principles can be applied to marine landscapes and artificial structures such as pier could already be function as corridors for both residential and migratory species.

Evidence was found that the piers are impacting on varying trophic levels within the ecosystem, from epibiota colonising the pilings through to predatory fish species such as Bass, which are feeding on the smaller prey species such as sand eel that are found in greater abundances around the piers. This research is the first to consider the food webs surrounding artificial structures and the findings show that a multilevel food web is operating around the piers which have changed the local distribution of mobile fauna.

The connectivity of marine populations is crucial to understand in order to conserve and maintain the marine ecosystem for future generations (Cowen, 2006). Without adequate connectivity between habitats biodiversity can be lost (Cowen et al., 2006). Artificial structures can either facilitate or prevent the connectivity of marine populations, they may act as stepping stones for species dispersal or alternatively they could create barriers (Forrest et al., 2009; Keith et al., 2011; Adams et al., 2014). In riverine systems artificial structures, such as weirs and dams, have shown to cause barriers to the dispersal and migration of fish species such as salmon and trout (Lucas & Frear, 1997; Ovidio & Philippart, 2002; Nilsson et al., 2005; Dudgeon et al., 2006). As a consequence, fish passes have been installed in certain areas to help fish populations overcome these obstacles (Baras et al., 1994; Lucas & Frear, 1997; Jansson et al., 2007).

Chapter 5 documented significant potential connectivity between natural and artificial habitats within Poole Bay and determined that the artificial structures within Poole Bay could act as stepping stones for species dispersal rather than barriers. Larvae with short pelagic larval durations will benefit to the greatest extent from well-connected structures as they only have the ability to travel small distances (Shanks et al., 2003; Shanks, 2009). The findings of this research support Adams et al., (2014) who discovered that marine renewable developments also have the potential to act as stepping stones in facilitating marine larvae, stressing the importance of habitat configuration and biogeography. Through the use of bio-physical models, potential invasion and/or expansion sites can be predicted based on the hydrodynamic flows of the area and these sites can then be monitored for any settlement of non-native species and managed if necessary.

8.3 Ecological Enhancement

Ecological engineering techniques have been used in terrestrial ecosystems for over 30 years (Mitsch, 2012), yet within the marine environment ecological enhancement experiments have only been conducted more recently, through small scale trials, in which the majority have been led by academics (Browne & Chapman, 2011; Firth et al., 2014b, 2016a; Coombes et al., 2015; Evans et al., 2015; Loke & Todd, 2016). In order to upscale the ecological enhancement techniques on a large scale, there is a need for acceptance by coastal managers, coastal engineers and policy makers. Engineers want proof that large scale projects will be successful but without the permission to implement designs on a large scale we cannot overcome this barrier. Retrofitted low cost options such as the holes and grooves described in Chapter 6 could have the potential to show engineers the benefits of ecological enhancement without a high amount of financial investment. Showcasing the ease and success of the ‘holes and grooves’ experiment may encourage engineers to try more complex built in designs in future projects. It has been acknowledged that improved documentation is needed in order to showcase the potential ecological enhancement techniques that could be implemented in existing and future coastal defence schemes (Evans, 2016; Sherrard, 2017). Existing literature for academic trials is typically in journal article format which might not be accessible for coastal mangers and engineers. Ideally new modes of communication need to be developed which enables practitioners to understand the ecological enhancement options available to them and ways in which they can obtain funding for the projects. Lack of funding is a common reason why ecological enhancement opportunities are missed or rejected as practitioners do not see it as a suitable investment. If we can overcome this hurdle by highlighting the secondary benefits of ecological enhancement such as community engagement and potential bio-protection we may be able to encourage more practitioners to use the techniques.

Improvements to legislation are required in order to improve the ecologically sensitive design of structures and create more opportunities for ecological enhancement. Current policy legislation as detailed in Chapter 1 Section 1.4.1, does not enforce ecological enhancement as a mandatory requirement in the construction of all coastal defence structures. It has been noted that there is a lack of long term monitoring (+ 24 months) for ecological enhancement trials of which are needed to drive policy change; this is predominantly due to a lack of funding and resources for monitoring. Without adequate funding, long term monitoring of ecological enhancement on artificial structures cannot be obtained which in turn limits the assessment of successes or failures.

This research has shown that simple low cost retrofitted options such as the holes and grooves in rock armour can have a significant impact on the community colonisation of coastal artificial structures. Increasing the surface texture and habitat heterogeneity created refuge from both environmental conditions and predation which in turn increased the species richness and diversity of the defence structure. Similar techniques have been used successfully in terrestrial environments, for example drilling holes for climbing plants in disused quarries (Wang et al., 2009) and using green rooves in urban environments (Grant & Lane, 2006). When presenting this research to the Institute of Civil Engineers (ICE) the feedback was very positive with several practitioners wanting to implement these techniques on their existing rock armour structures. It is important to note that not all enhancements will be suitable to all environments; firstly the availability of suitable larvae is key, as a structure will not become colonised by marine species if there are no nearby populations available to produce larvae. This is particularly important when enhancing a structure for a specific species (e.g. oysters) because if there are no local populations the attempt with be a failure. Additionally the wave energy, tidal range and sediment type are all important factors which will affect the success of ecological enhancements. Attaching precast concrete devices in wave exposure habitats may result in damage to enhancement devices and or coastal structures and using techniques at incorrect tidal heights or within inappropriate habitats may also lead to unsuccessful results. Existing knowledge and examples need to be consolidated in order to provide a useful guide for policy makers and practitioners.

8.4 Future work

Although this thesis has addressed existing knowledge gaps, it has also exposed further gaps in understanding which need to be explored in future studies. Firstly the extent of artificial structures around the UK needs to be mapped in order to obtain accurate details on location, type and coverage of coastal structures around the UK. Whilst this has been conducted for sections of the UKs coastline there are significant gaps in the knowledge within some regions. As mentioned previously, a comprehensive guide needs to be created detailing ways in which ecological enhancement can be incorporated into coastal defence structures alongside an assessment of the secondary benefits which could be created through their installation. A key aspect of this will be suggesting potential funding sources for both the implementation of ecological enhancement and the post-construction monitoring of both structural integrity and ecological communities.

Additional work to assess the long term variation in mobile communities associated with artificial structure is important and tracking studies could be conducted to identity areas which are visited regularly by both resident and migratory species in order to highlight areas of conservation value. Genetic studies could also be applied to determine the connectivity of spatially separate populations and the potential for artificial structures to act as stepping stones (Sammarco et al., 2012).

Conclusion

Ocean sprawl is occurring throughout the world and coastal ecosystems are being altered through the construction of hard artificial structures (Firth et al., 2016a; Bishop et al., 2017). This thesis focused on improving the understanding of the ecology of artificial structures in order to enhance communities associated with structures and prevent the loss of biodiversity. The results will aid practitioners in the management of current structures and in the construction of future projects. Complex and heterogeneous structures should be created where possible in order to encourage and maintain biodiversity both on and around artificial structures. This work has shown evidence that artificial structures can have an impact on species at varying trophic levels, therefore the wider implications need to be understood for effective conservation and management. The consequence of building further artificial structures around ports and harbours or within protected areas should be considered carefully and if possible modelled to assess the extent in which new structures will connect with existing natural and artificial habitats within the region. Lessons can be learnt from studies around the world, yet it is important to remember that each location is different and the impact and enhancement of artificial structures needs to be considered on a case by case basis in order to produce results suitable for each location and, as with natural rocky shores, variation is inevitable. Future marine planning needs to involve collaborations between scientists, policy makers and practitioners in order to combine research, expertise and knowledge on the marine environment as a whole and work together to provide ecologically enhanced marine infrastructure.

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Appendix 1

Table A1.1: Generalised linear mixed models (GLMMs) of species diversity in relation to environmental variables on natural and artificial shore within four regions on the south coast of England (df= degrees of freedom, AIC= Akaike's Information Criterion, Δ AIC= difference in AIC from the most suitable model).

a) Species Richness

	GLMM (Fixed Effects)	**df**	**Log-likelihood**	**AIC**	**Δ AIC**
A	Substrate	7	-451.92	917.85	169.62
B	Distance	7	-400.22	814.44	66.21
C	Zone	6	-398.69	809.35	61.12
D	Distance+ Zone	8	-392.33	800.66	52.43
E	**Substrate + Distance + Zone**	**10**	**-364.11**	**748.23**	**0**

b) Total Abundance

	GLMM (Fixed Effects)	**df**	**Log-likelihood**	**AIC**	**Δ AIC**
A	Substrate	7	-1195.2	2404.5	
B	Distance	7	-1177.3	2368.7	
C	Zone	6	-1171.7	2355.3	
D	Distance+ Zone	8	-1194.7	2405.4	
E	Substrate + Distance + Zone	10	1189.1	2368.2	

c) Species Diversity (H)

	GLMM (Fixed Effects)	**df**	**Log-likelihood**	**AIC**	**Δ AIC**
A	Substrate	7	-340.49	694.99	77.08
B	Distance	7	-327.30	668.61	50.70
C	Zone	6	-321.26	654.51	36.60
D	Distance+ Zone	8	-311.24	638.48	20.57
E	**Substrate + Distance + Zone**	**10**	**-298.95**	**617.91**	**0**

Table A1.2: ANOVA results of the GLMMs for a) Species richness, b) Total abundance, c) Species diversity (***= P<0.001, *=P<0.05, NS= P>0.05).

a) Species Richness (S)				
Source	**Sum Sq**	**Mean Sq**	**F**	**P**
Substrate	9.55	4.77	29.11	***
Distance	7.63	3.81	23.26	***
Zone	9.52	9.56	58.26	***
b) Total Abundance (N)				
Source	**Sum Sq**	**Mean Sq**	**F**	**P**
Substrate	22.43	11.21	5.61	*
Distance	8.72	4.36	2.18	NS
Zone	56.88	56.88	28.49	***
c) Species Diversity (H)				
Source	**Sum Sq**	**Mean Sq**	**F**	**P**
Substrate	10.17	5.08	37.33	***
Distance	2.84	1.42	10.44	***
Zone	4.41	4.41	32.39	***

Table A1.3: Generalised linear mixed models (GLMMs) of species diversity in relation to environmental variables within four regions on the south coast of England (df= degrees of freedom, AIC= Akaike's Information Criterion, Δ AIC= difference in AIC from the most suitable model).

a) Species Richness
lmer(sqrt(S)~Substrate+Aspect+Distance+Zone+Distance*Zone+Aspect*Distance +1|Region/Subs_TG), data=datav2)

	GLMM (Fixed Effects)	df	Log- likelihood	AIC	Δ AIC
A	Aspect+ Distance + Zone	8	-114.56	245.11	32.29
B	Substrate + Aspect + Zone	9	-105.39	228.78	15.96
C	Substrate + Aspect + Zone + Distance*Zone	11	-99.49	220.99	8.17
D	**Substrate + Aspect + Zone + Distance*Zone + Aspect*Distance**	**13**	**-93.40**	**212.82**	**0**

b) Total Abundance
lmer(LogeN~Substrate+Aspect+Distance+Zone +(1|Region/Subs_TG),data=datav2)

	GLMM (Fixed Effects)	df	Log- likelihood	AIC	Δ AIC
A	Aspect + Distance + Zone	8	-594.38	1204.8	2.3
B	**Substrate + Aspect + Distance + Zone**	**9**	**-592.25**	**1202.5**	**0**

c) Species Diversity (H)
lmer(H~Substrate+Aspect+Distance+Zone+Distance*Zone+Aspect*Distance+Substrate *Aspect +(1|Region/Subs_TG), data=datav2)

	GLMM (Fixed Effects)	df	Log- likelihood	AIC	Δ AIC
A	Aspect+ Distance + Zone	8	-118.83	253.65	37.38
B	Substrate + Aspect + Zone	9	-110.62	239.25	22.98
C	Substrate + Aspect + Zone + Distance*Zone	11	-104.92	231.84	15.57
D	Substrate + Aspect + Zone + Distance*Zone + Aspect*Distance	13	-96.17	218.34	2.07
E	**Substrate + Aspect + Zone + Distance*Zone + Aspect*Distance + Substrate*Aspect**	**14**	**-94.13**	**216.27**	**0**

Table A1.4: ANOVA results of the GLMMs for a) Species richness, b) Total abundance, c) Species diversity of wooden and rock groynes (***= P<0.001, *=P<0.05, NS= P>0.05).

a) Species Richness (S)

Source	**Sum Sq**	**Mean Sq**	**F**	**P**
Substrate	8.7706	8.7706	91.372	***
Aspect	0.2376	0.2376	2.475	NS
Distance	2.5015	1.2507	13.030	***
Zone	0.1261	0.1261	1.314	NS
Distance*Zone	0.5140	0.2570	2.678	NS
Aspect*Distance	1.1192	0.5596	5.830	**

b) Total Abundance (N)

Source	**Sum Sq**	**Mean Sq**	**F**	**P**
Substrate	21.322	21.322	12.1229	***
Aspect	3.192	3.192	1.8145	NS
Distance	10.014	5.007	2.8466	NS
Zone	33.997	33.997	19.3292	***

c) Species Diversity (H)

Source	**Sum Sq**	**Mean Sq**	**F**	**P**
Substrate	9.0385	9.0385	93.524	***
Aspect	0.3861	0.3861	3.995	*
Distance	1.2379	0.6190	6.405	**
Zone	0.0126	0.0126	0.130	NS
Distance*Zone	0.4920	0.2460	2.545	NS
Aspect*Distance	1.4433	0.7216	7.467	***
Substrate*Aspect	0.3148	0.3148	3.257	NS

Appendix 2

Table A2.1: Discrimination factors for blood and muscle from consumers

Species	C	N	Source
D. labrux	0.4 (+/- 0.1)	3.4 (+/-0.1)	Post (2002)
T. luscus	0.4 (+/- 0.1)	3.4 (+/-0.1)	Post (2002)
S. solea	-0.5 (+/- 0.1)	4.5 (+/- 0.1)	-
O. eperlanus	0.5	2.4	Hansson et al (1997) Vizzini & Mazzola (2002)
A. tobianus	0.5	2.4	Hansson et al (1997) Vizzini & Mazzola (2002)

Table A2.2: 2-way ANOVA for comparison of benthic infauna Species richness (S), Total abundance (N) and Species diversity (H) between Distance away from the pier (1m, 10m, 25m) and Distance along the pier (Inner, Middle, Outer) at a) Bournemouth and b) Boscombe Pier (***=P<0.001, **=P<0.01,*=0.05 NS= Not significant).

a) Bournemouth Pier

	S			N			H		
Source of Variation	df	F	p	df	F	p	df	F	p
Distance away (DAw)	2	1.144	NS	2	0.29	NS	2	0.68	NS
Distance along (DAl)	2	0.87	NS	2	1.51	NS	2	0.01	NS
DAw x DAl	4	0.26	NS	4	0.08	NS	4	0.84	NS
Residuals	18			18			18		

b) Boscombe Pier

	S			N			H		
Source of Variation	df	F	p	df	F	p	df	F	p
Distance away (DAw)	2	0.72	NS	2	1.10	NS	2	0.40	NS
Distance along (DAl)	2	0.98	NS	2	1.76	NS	2	0.40	NS
DAw x DAl	4	0.81	NS	4	0.68	NS	4	0.54	NS
Residuals	18			18			18		

Figure A2.1: Images of a) Bournemouth Pier and b) Boscombe Pier to illustrate variation in structural complexity.

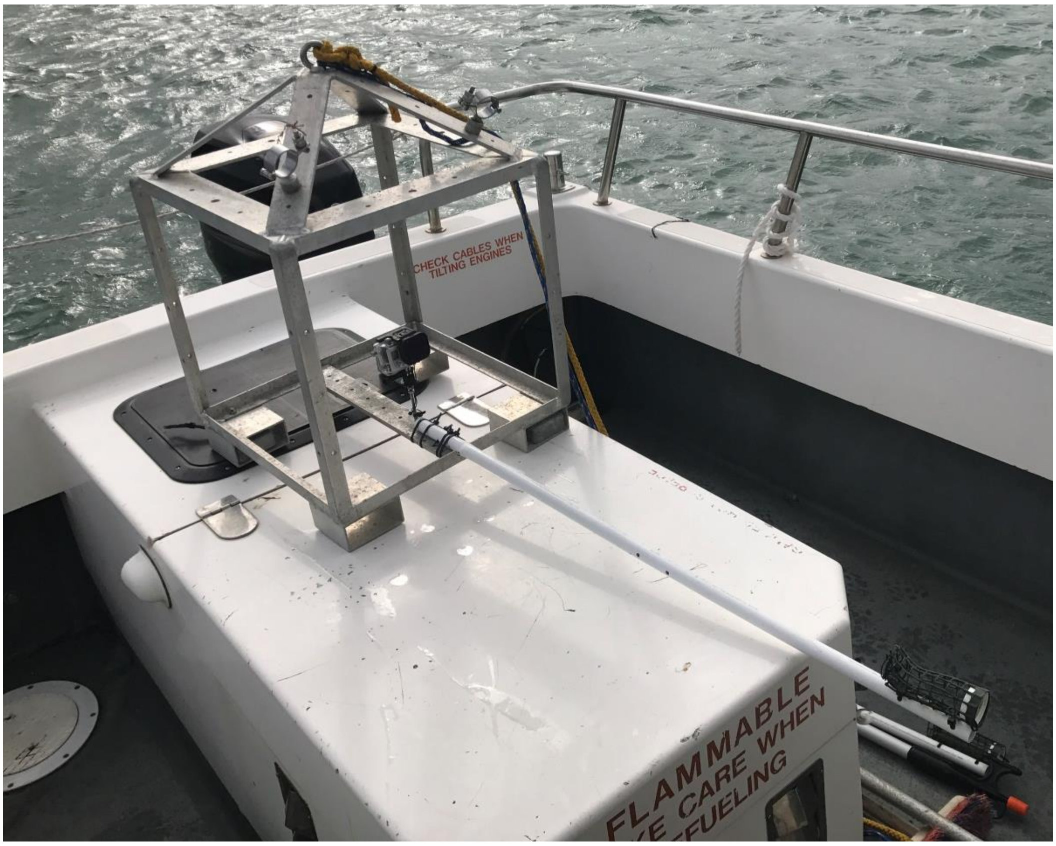

Figure A2.2: Image of the Baited Remote Underwater Video (BRUV) deployment unit.

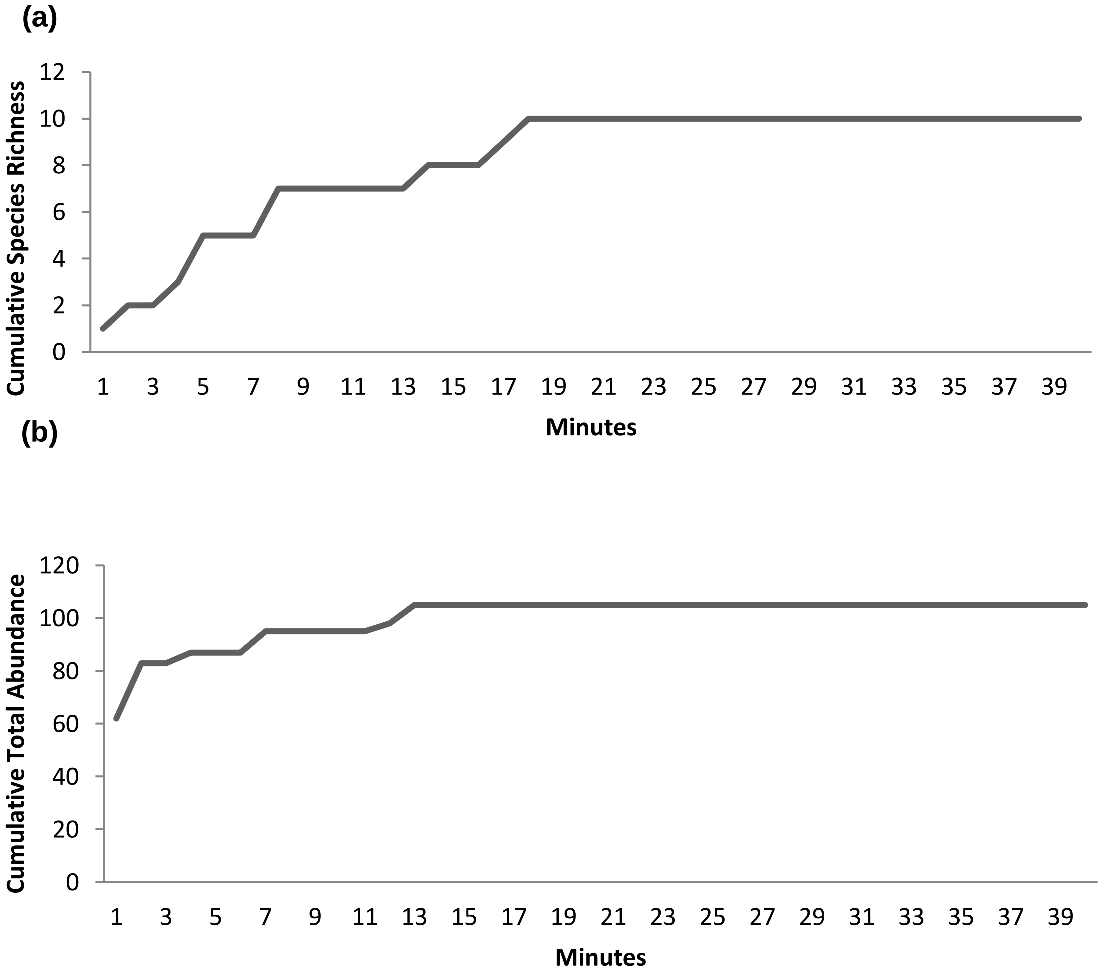

Figure A2.3: (a) Cumulative species richness for a single 40 minute deployment at 1m, (b) Cumulative Total abundace for 40minute deployment at 1m off Bournemouth Pier, June 2015. Maximum species richness (MaxD) and maximum total abundance (MaxN) were achieved after 18 and 13 mins resepctively.

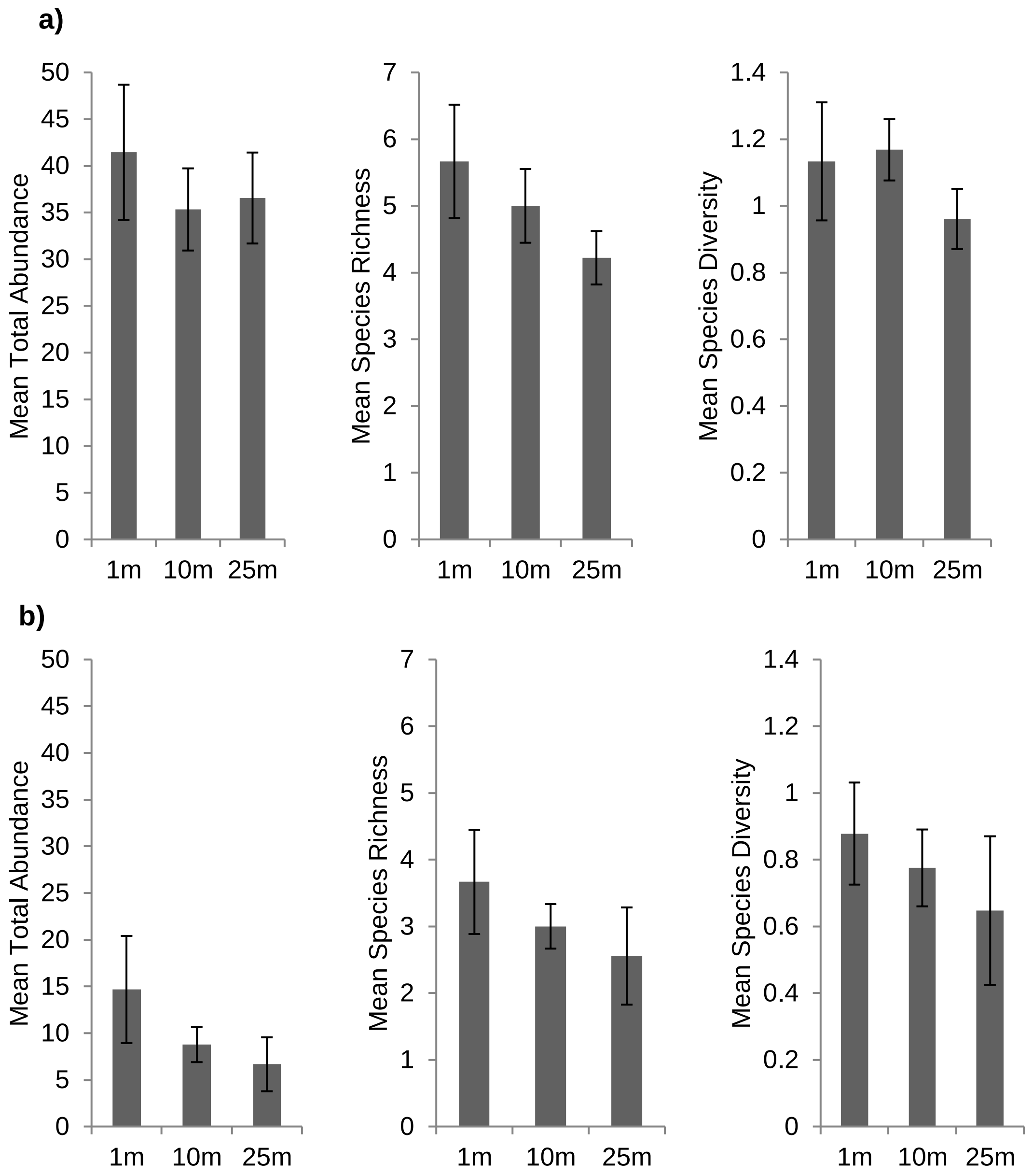

Figure A2.4: Mean Species richness, total abundance and species diversity of benthic infauna with varying distance away from the pier at a) Bournemouth Pier and b) Boscombe Pier (+/- S.E)

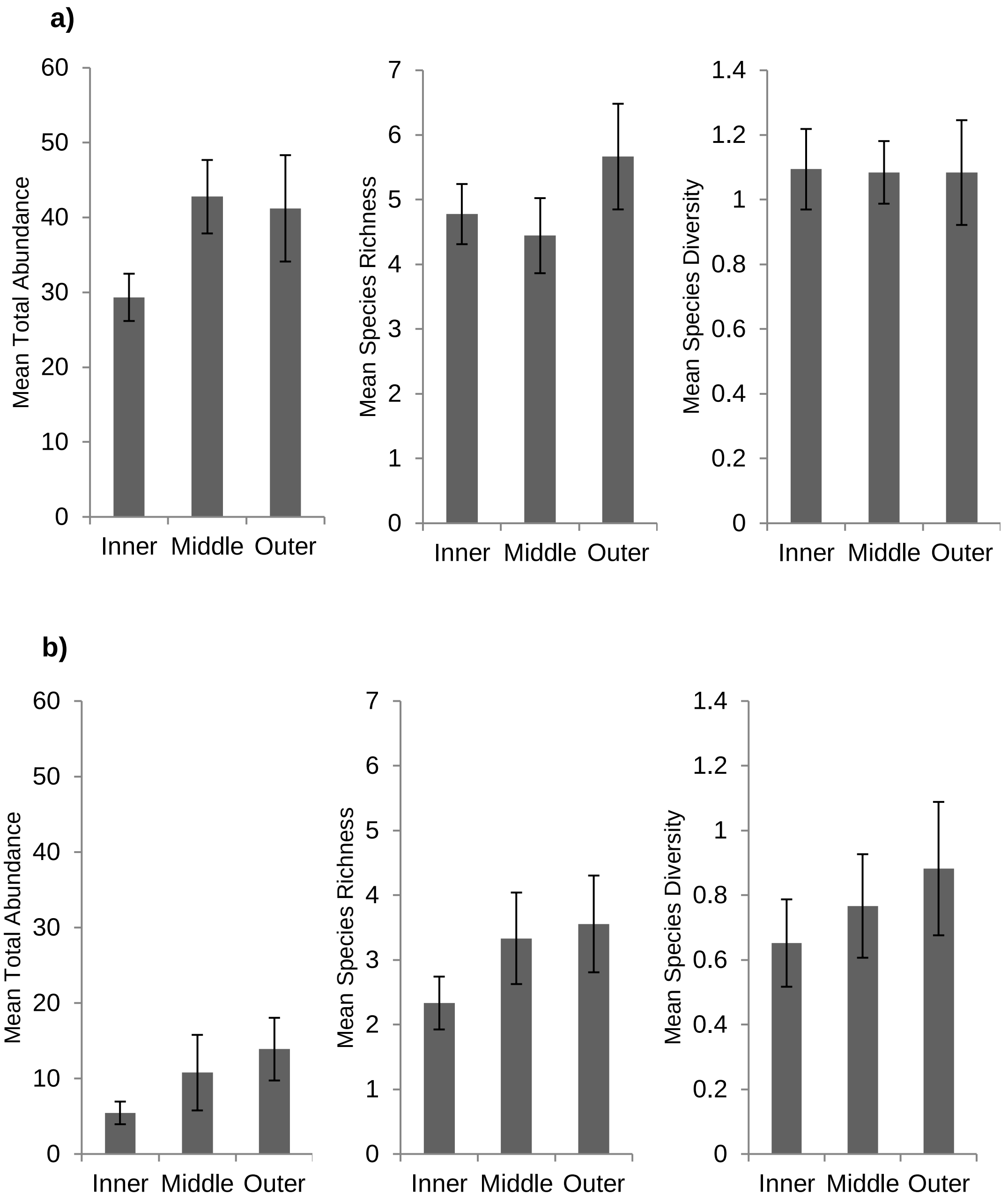

Figure A2.5: Mean Species richness, total abundance and species diversity of benthic infauna with varying distance along the pier at a) Bournemouth Pier and b) Boscombe Pier (+/- S.E)

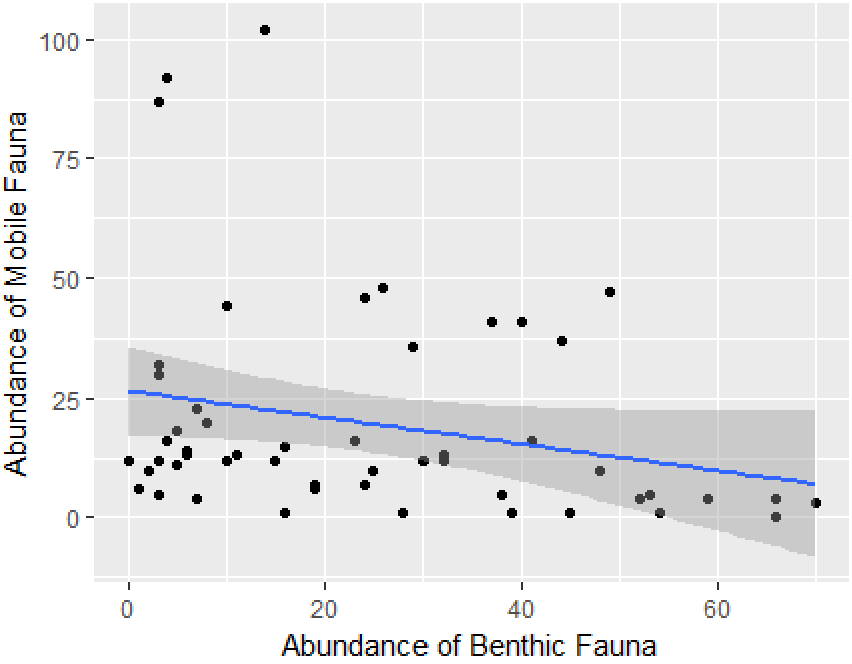

Figure A2.6: Correlation between the abundance of mobile fauna and the abundance of benthic fauna recorded around the piers.

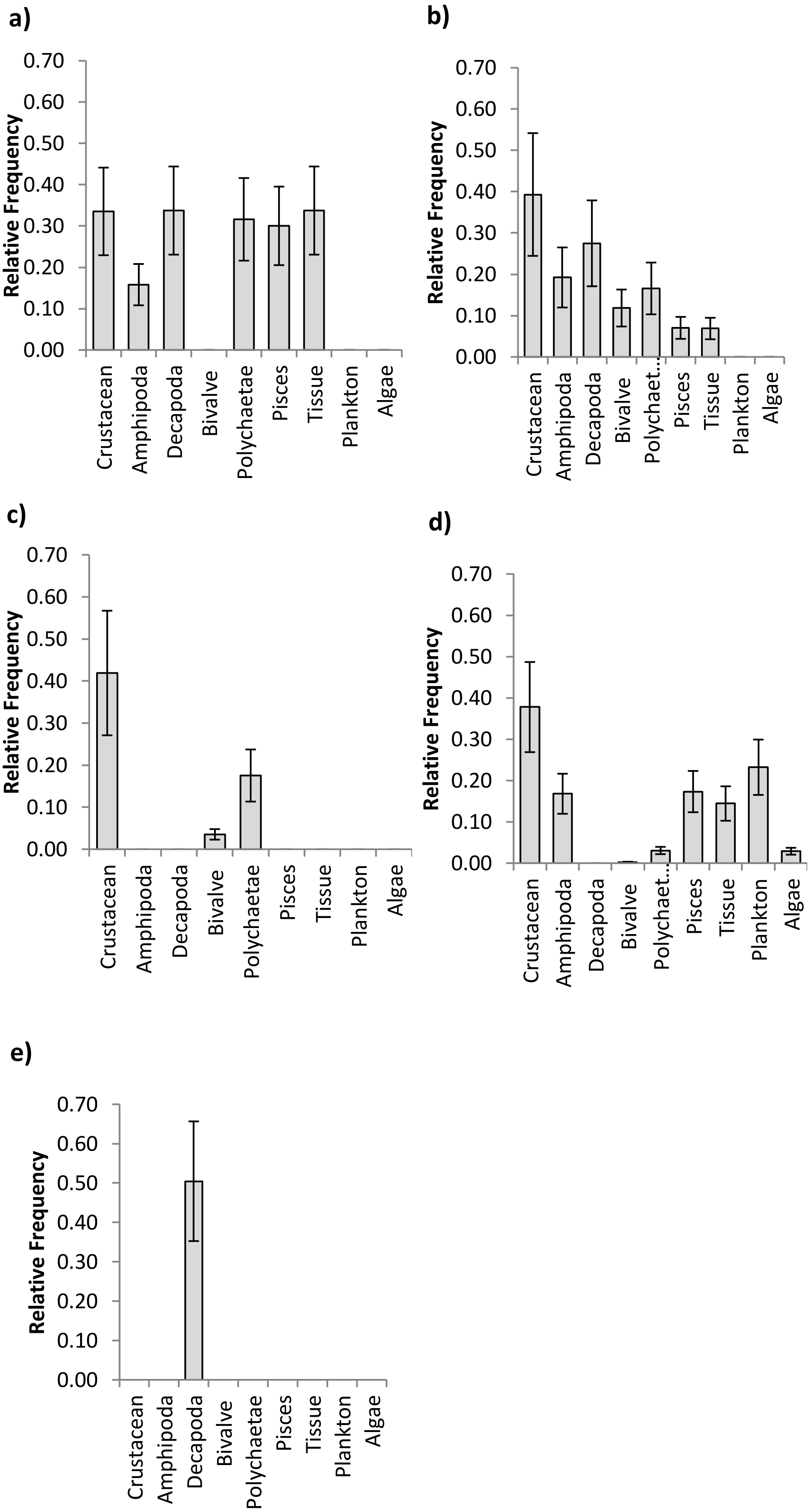

Figure A2.7: Stomach content relative frequency of prey categories for a) D. labrux b) T. luscus and c) S. solea d) O. eperlanus e) A. tobianus

Appendix 3

TELEMAC-2D Model details

To produce the hydrodynamic model for Poole Bay, bathymetric data of the region was obtained from HR Wallingford, from which a variable density triangulation mesh was created and interpolated using BlueKenue (NRC, 2016). In order to run the TELEMAC-2D model there was a steering file, boundary condition file, geometry file and a liquid boundary file. The steering file is the parameter file which controls the computation of the model. The boundary condition file contains information on all points on a boundary line and details whether or not they are open or closed boundaries. The geometry file contains information on the mesh including number of nodes, elements and the coordinates of each node. The liquid boundary files allows you to specify values for boundary conditions such as tracers, velocity and depth.

Larval dispersal model details

At each time step the water velocity was combined with the larvae model and the larvae were moved using the Runge Kutta 4th order integration iterative method to re-position the larvae (North et al., 2009). The particles new position was checked for model integrity to check if the larvae was within the model domain and not on dry land or out of the boundary. If the new position was valid then the larval position was renewed and if not the larvae were removed from the model. Once the new position had been validated the output data were recorded in terms of larval hits in designated boxes if required. Any larvae which crossed the wet boundaries of the model were removed and not allowed to return to the model (North et al., 2009).

The larvae were assumed to have no swimming capabilities and their location was determined by the velocity at each time step. To prevent inaccurate aggregation in the model due to diffusivity a variable time stepping scheme was used with sub-time steps

(5s) for vertical dispersion coupled with longer time steps for the horizontal dispersion (50s) (Mead, 2008; Herbert et al., 2012). A coefficient diffusivity of 0.02 m^2s^{-1} in the horizontal and 0.001 m^2s^{-1} in the vertical were used (Fischer et al., 1979). Wind and wave forcing were not modelled in order to allow the effect of tidal forcing alone to be observed.

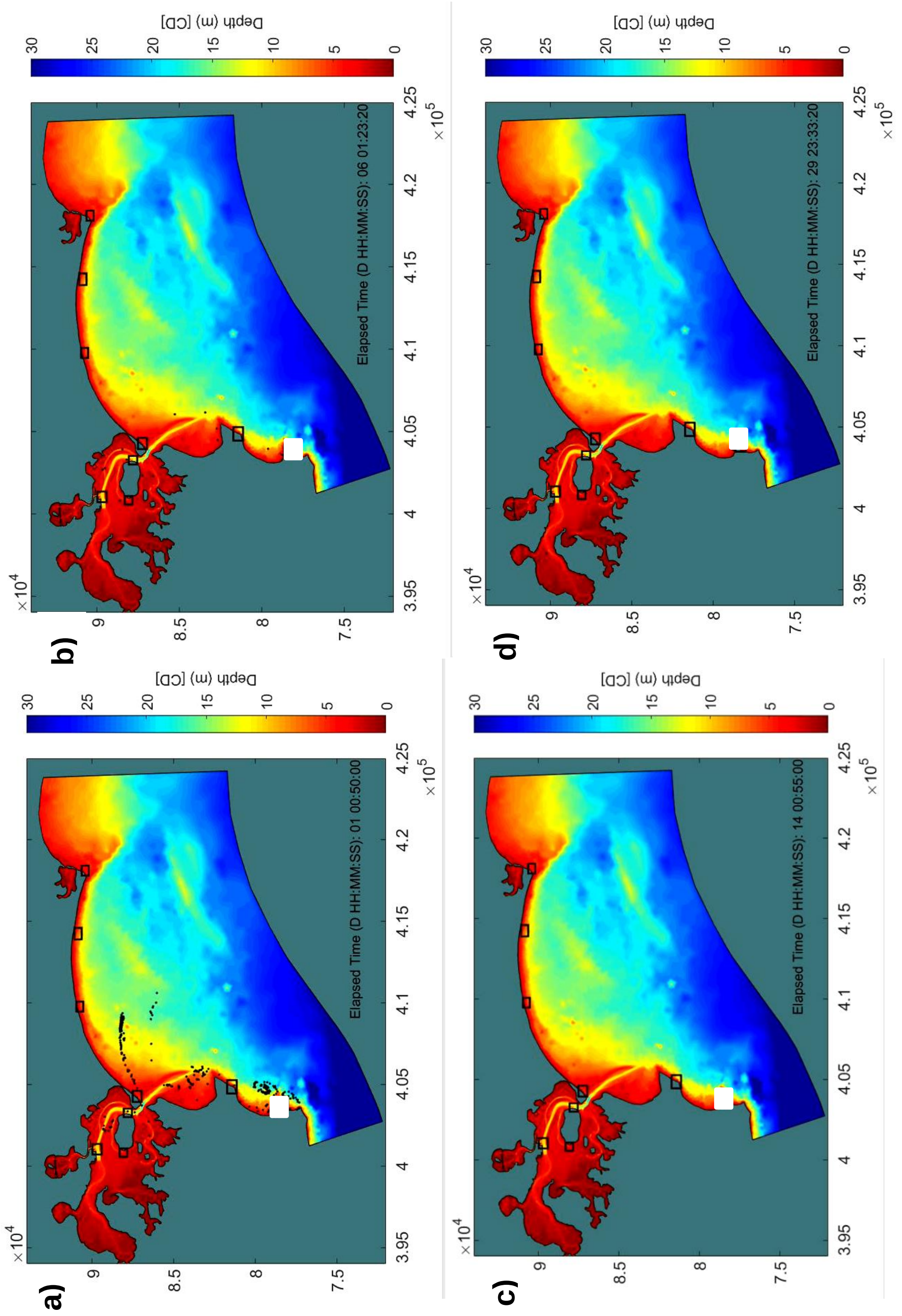

Figure A3.1 Modelled larvae dispersal from Site 1 Peveril Point for varying pelagic larval dispersal times a) 24 hours, b) 6 days, c) 14 days and d) 30 days. White box indicates source location, black boxes indicate destinations and black dots represent larvae.

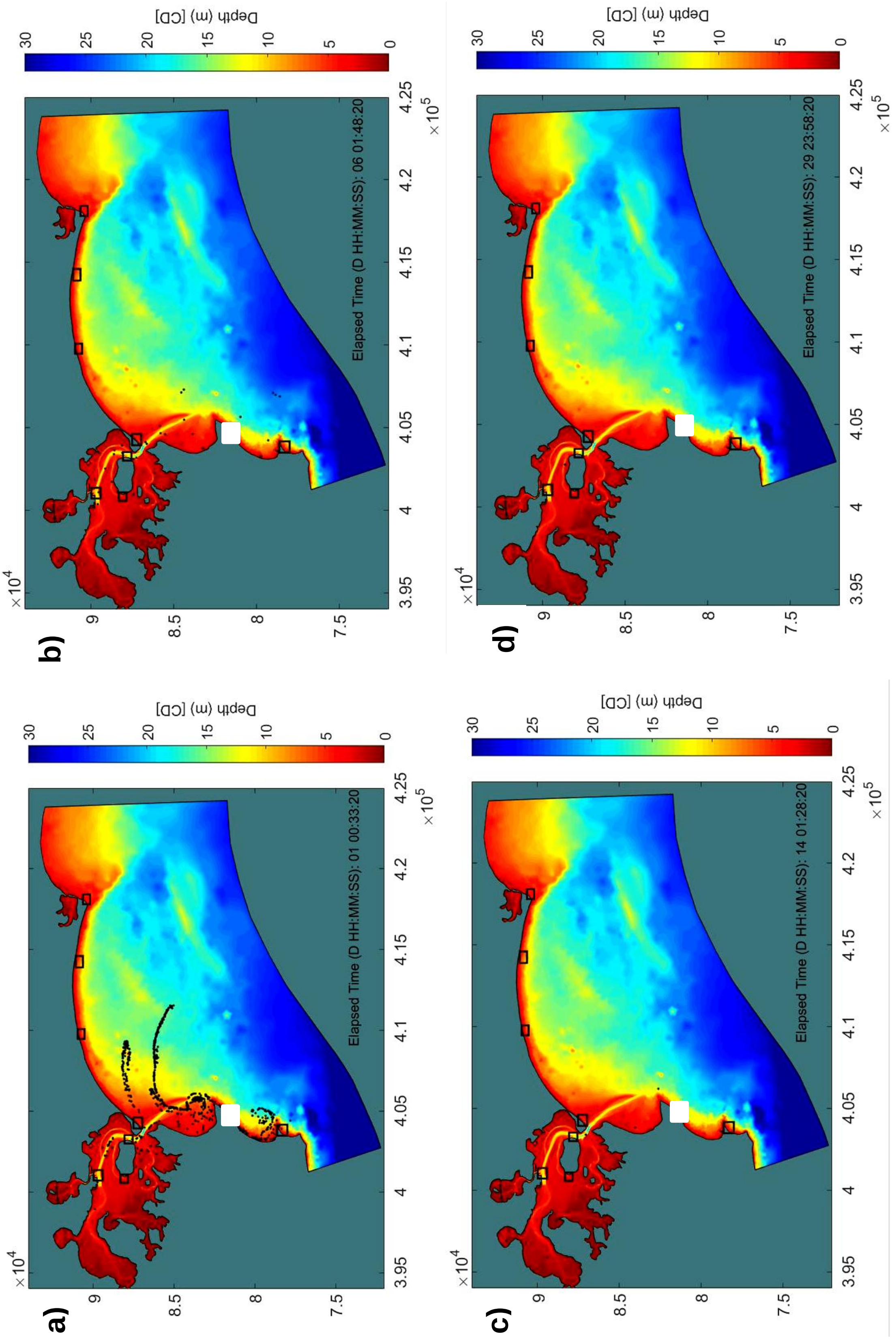

Figure A3.2 Modelled larvae dispersal from Site 2 Swanage for varying pelagic larval dispersal times a) 24 hours, b) 6 days, c) 14 days and d) 30 days. White box indicates source location, black boxes represent destination boxes and black dots represent larvae.

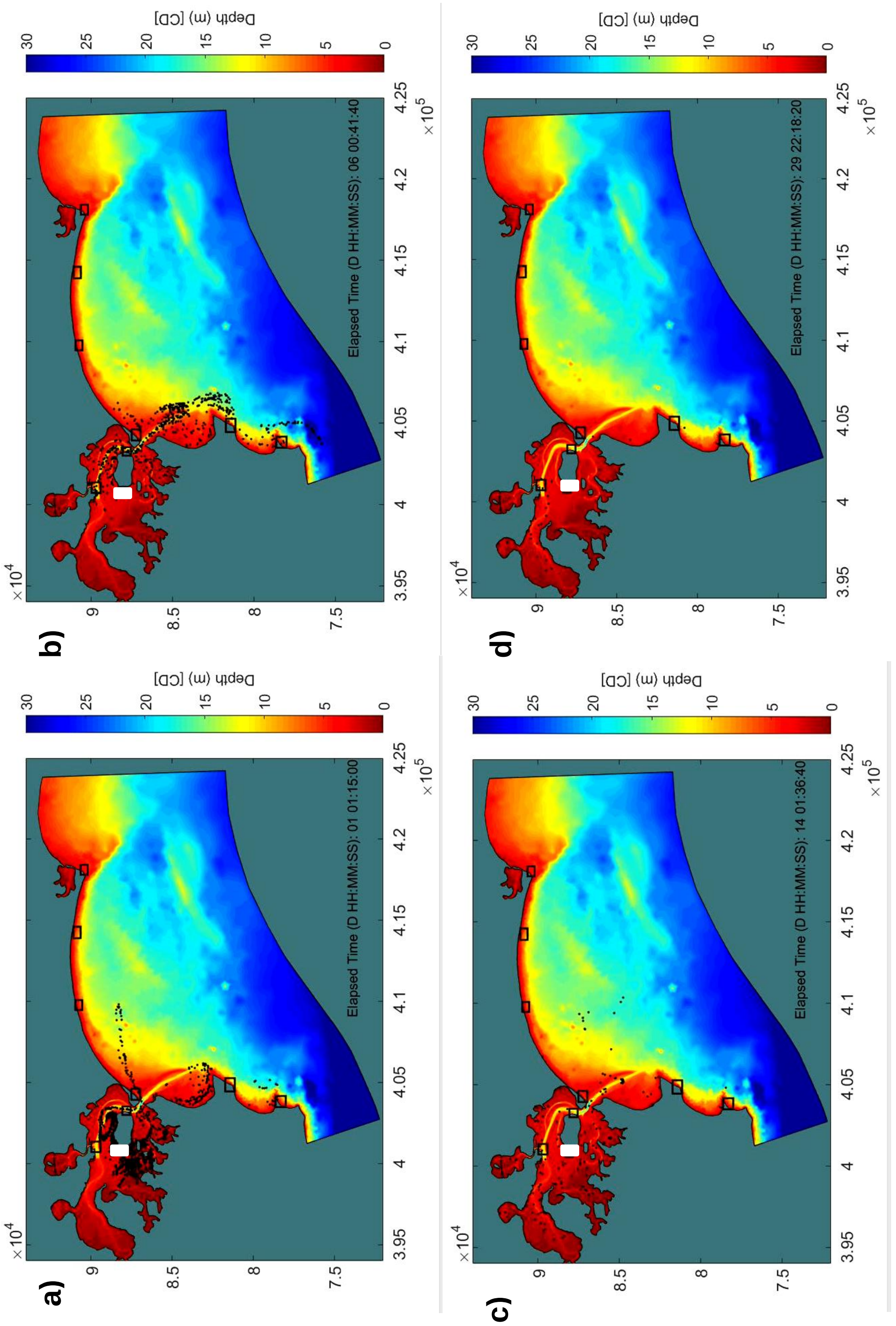

Figure A3.3 Modelled larvae dispersal from Site 3 Brownsea Pier for varying pelagic larval dispersal times a) 24 hours, b) 6 days, c) 14 days and d) 30 days. White box indicates source location, black boxes indicate destinations and black dots represent larvae.

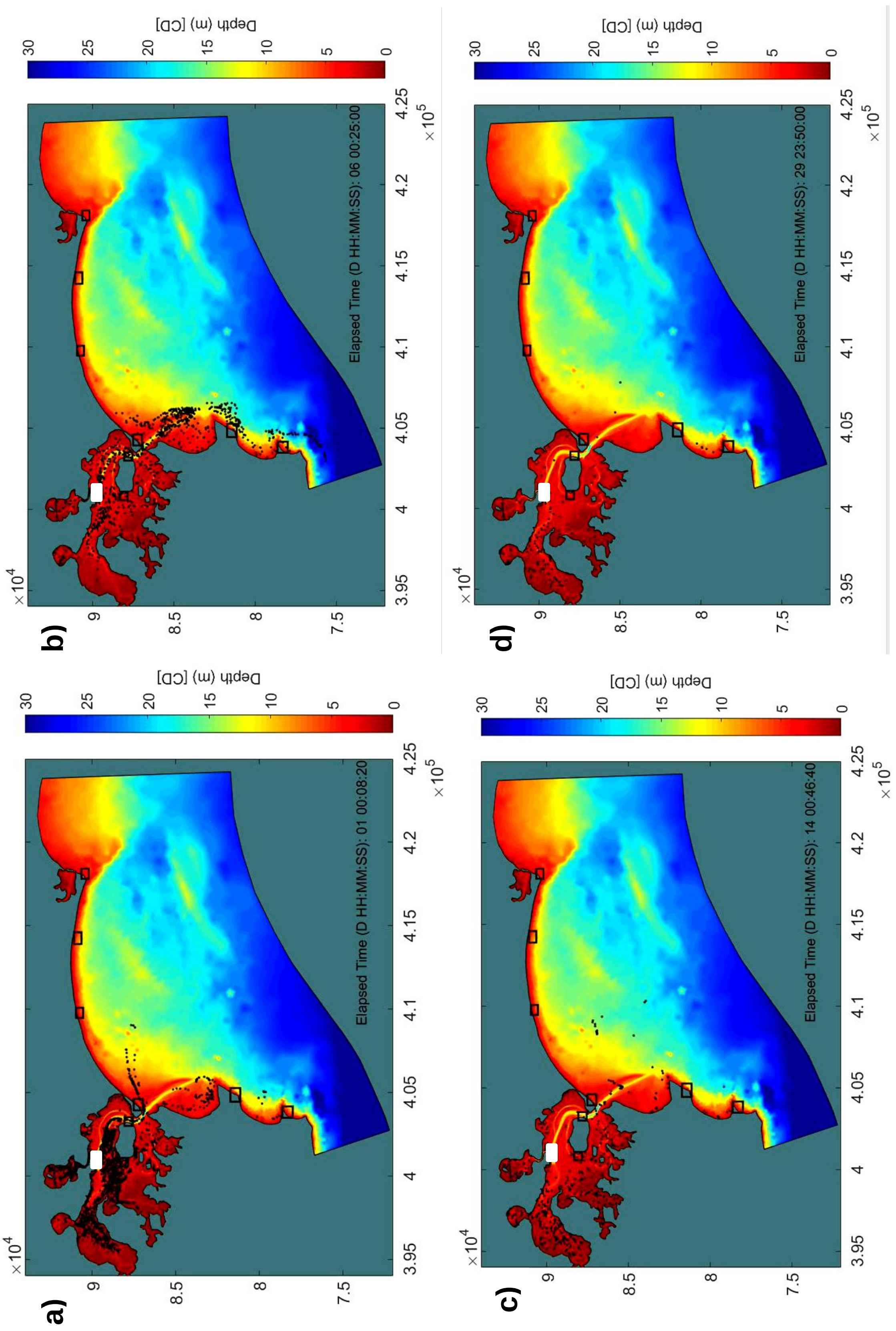

Figure A3.4 Modelled larvae dispersal from Site 4 Poole Port for varying pelagic larval dispersal times a) 24 hours, b) 6 days, c) 14 days and d) 30 days. White box indicates source location, black boxes indicate destinations and black dots represent larvae.

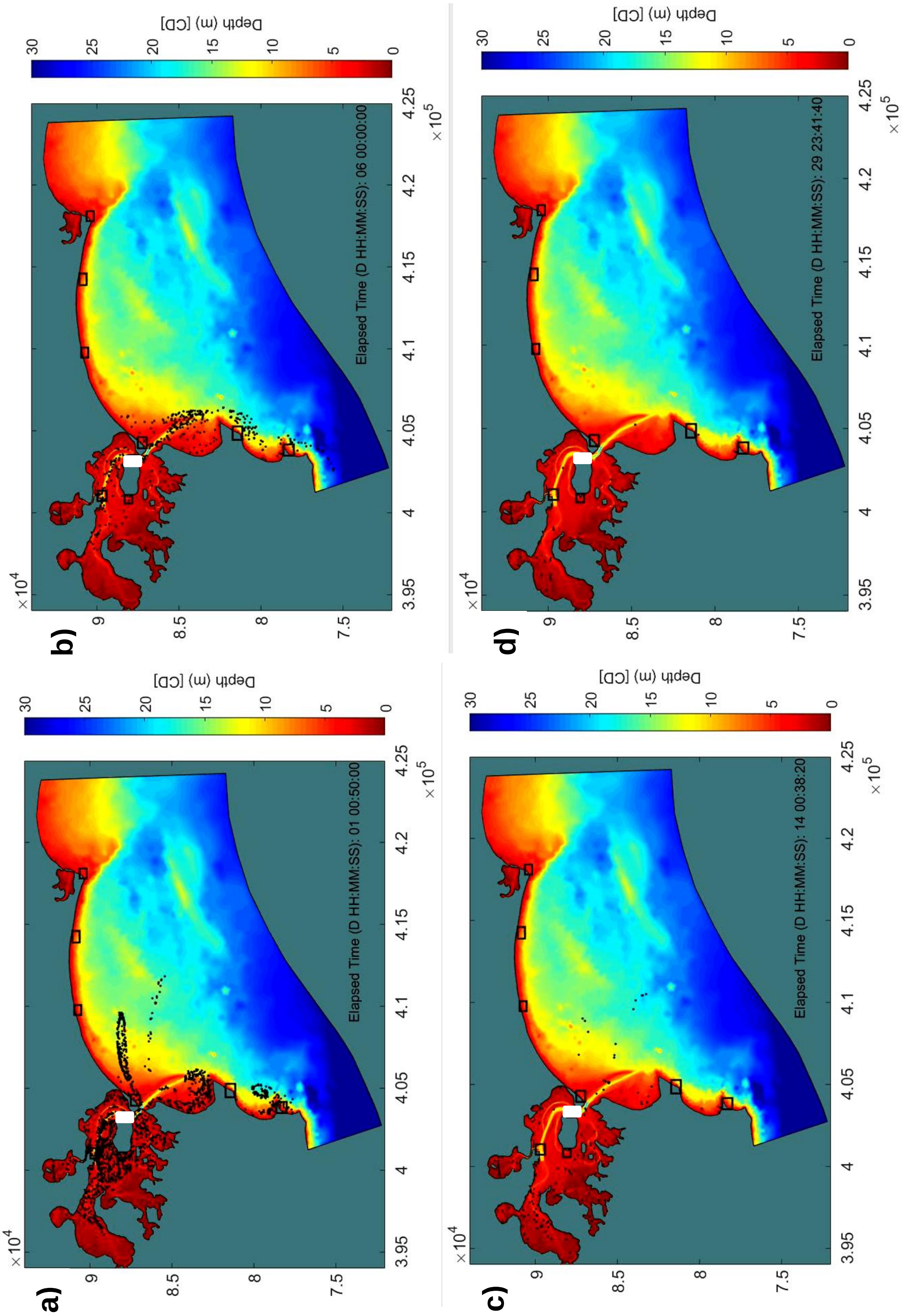

Figure A3.5 Modelled larvae dispersal from Site 5 Castle Pier for varying pelagic larval dispersal times a) 24 hours, b) 6 days, c) 14 days and d) 30 days. White box indicates source location, black boxes indicate destinations and black dots represent larvae.

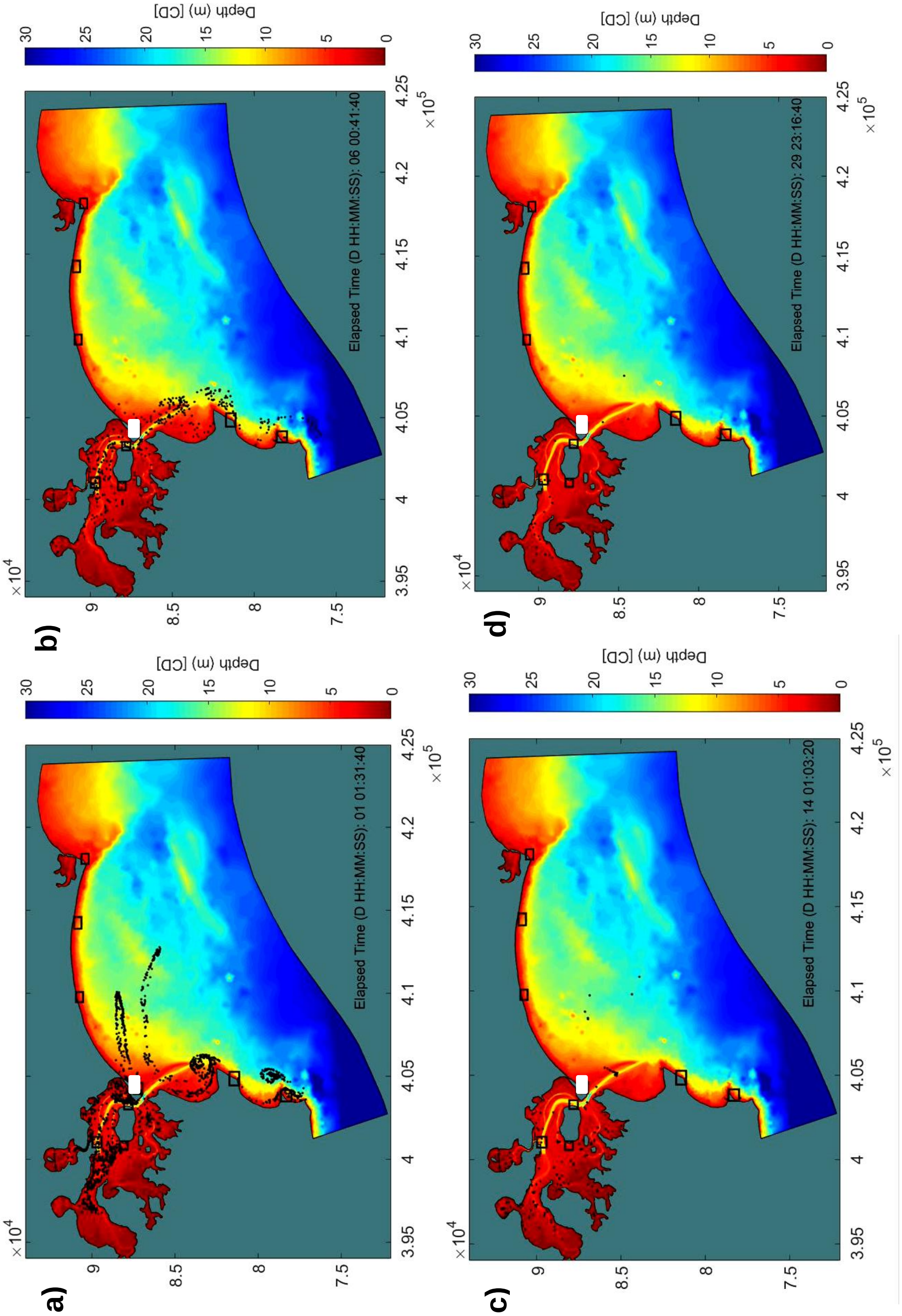

Figure A3.6 Modelled larvae dispersal from Site 6 Sandbanks for varying pelagic larval dispersal times a) 24 hours, b) 6 days, c) 14 days and d) 30 days. White box indicates source location, black boxes indicate destinations and black dots represent larvae.

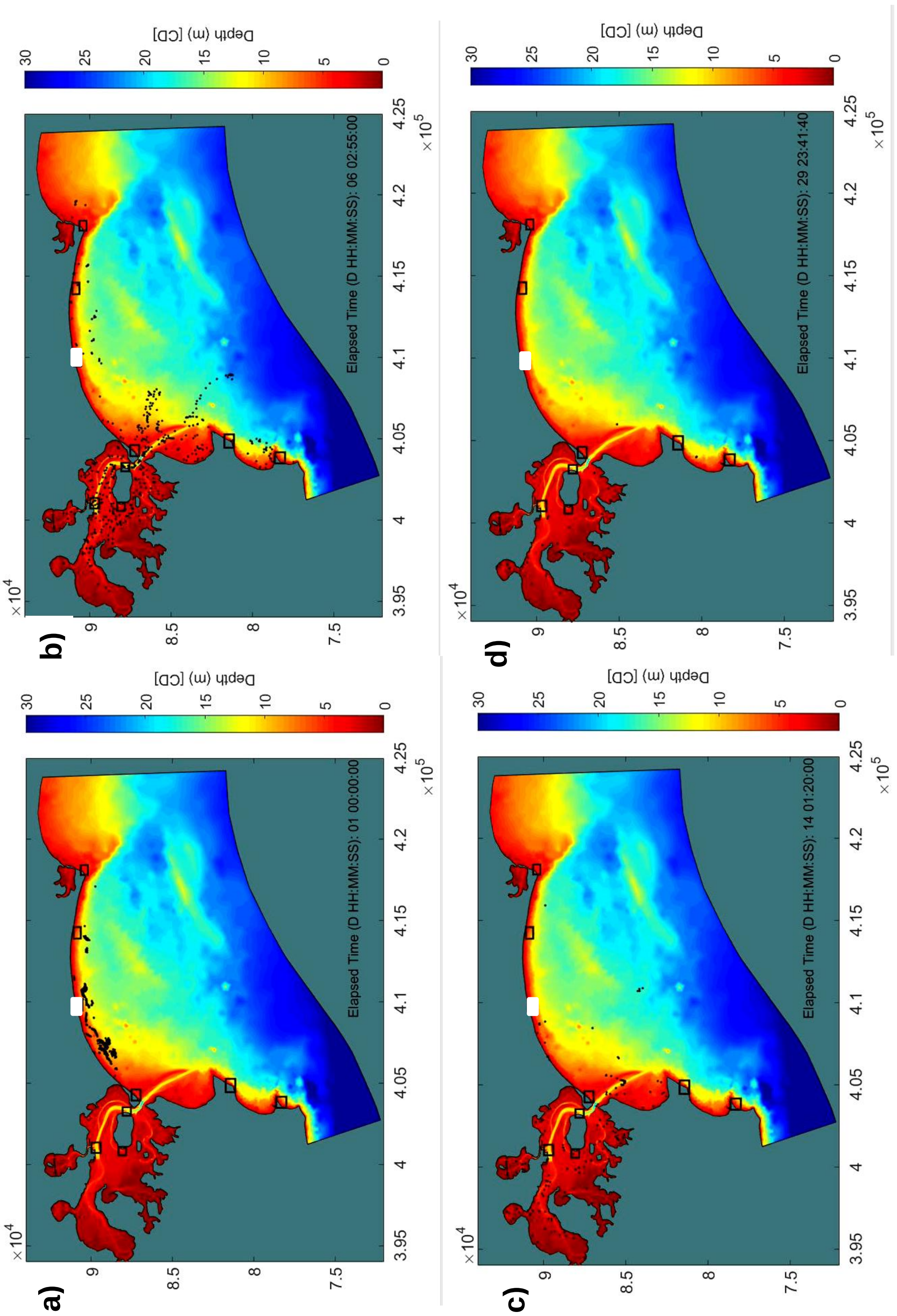

Figure A3.7 Modelled larvae dispersal from Site 7 Boscombe for varying pelagic larval dispersal times a) 24 hours, b) 6 days, c) 14 days and d) 30 days. White box indicates source location, black boxes indicate destinations and black dots represent larvae.

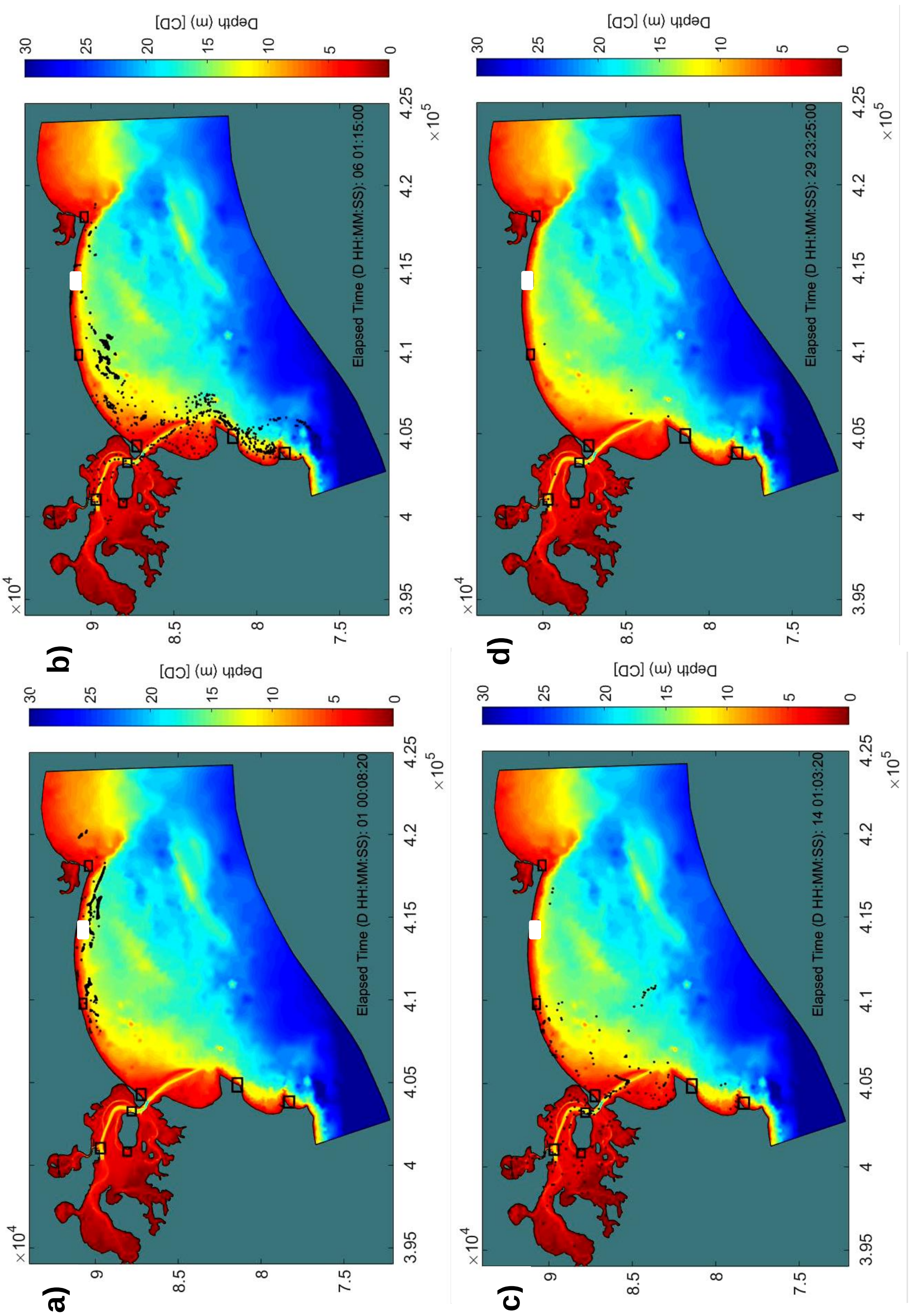

Figure A3.8 Modelled larvae dispersal from Site 8 Hengistbury for varying pelagic larval dispersal times a) 24 hours, b) 6 days, c) 14 days and d) 30 days. White box indicates source location, black boxes indicate destinations and black dots represent larvae.

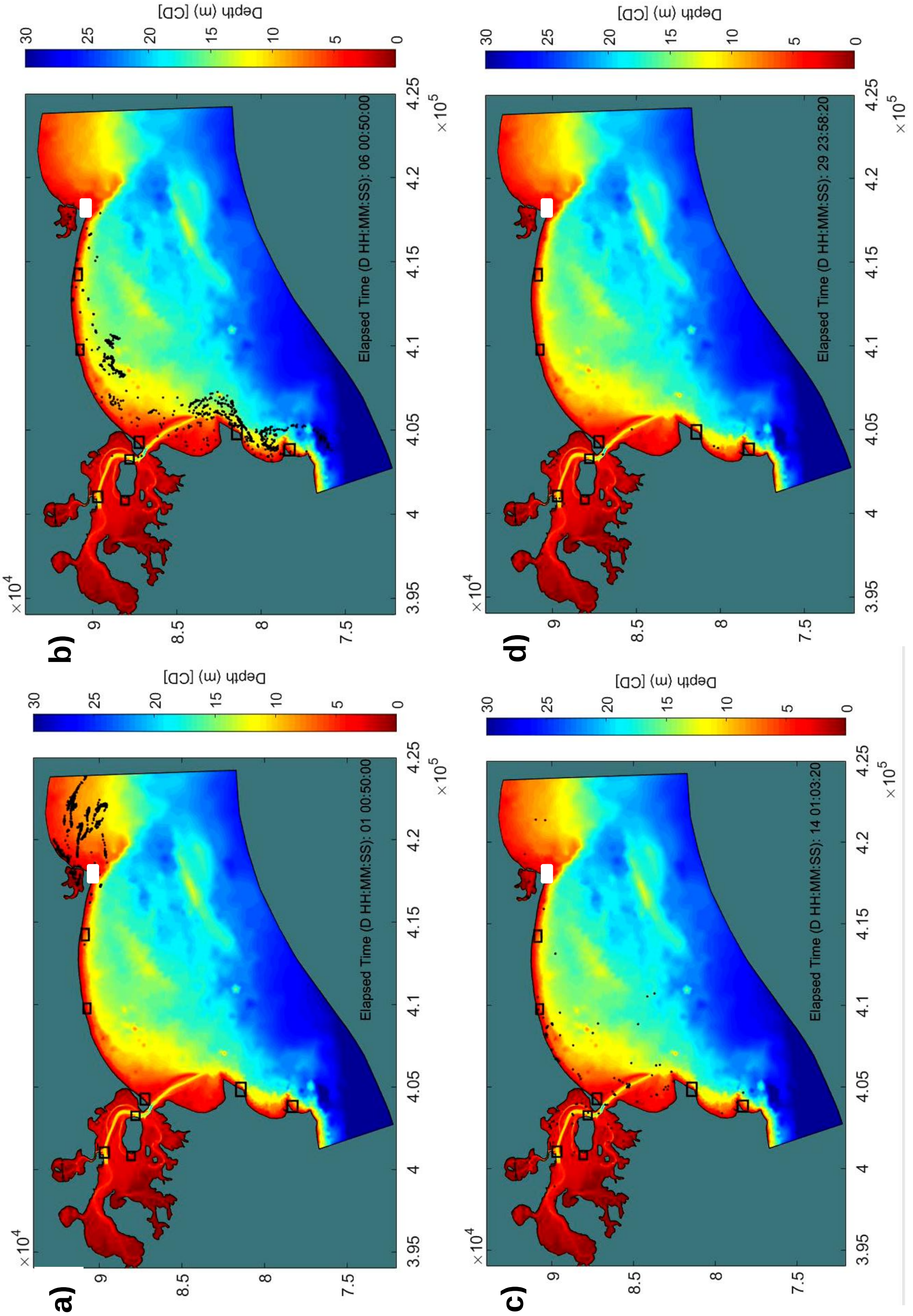

Figure A3.9 Modelled larvae dispersal from Site 9 Long Groyne for varying pelagic larval dispersal times a) 24 hours, b) 6 days, c) 14 days and d) 30 days. White box indicates source location, black boxes indicate destinations and black dots represent larvae.

Table A3.1 Mean number of initial 2000 particles within each patch after 24hours (Patch and Source: 1= Peveril, 2=Swanage, 3= Brownsea, 4=Poole Port, 5=Castle Pier, 6=Sandbanks, 7=Boscombe, 8=Hengistbury, 9=Long Groyne).

		Destination								
		1	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**
Source	**1**	65.90 (35.41)	23.92 (45.53)	0.00	0.00	0.89 (1.98)	0.04 (0.08)	0.00	0.00	0.00
	2	1.48 (1.31)	84.33 (46.86)	36.12 (80.77)	0.82 (1.82)	4.72 (4.84)	0.18 (0.18)	0.00	0.00	0.00
	3	1.00 (2.23)	0.64 (0.77)	139.81 (78.38)	12.98 (22.68)	20.72 (26.98)	39.52 (88.37)	0.00	0.00	0.00
	4	3.56 (7.96)	1.52 (3.40)	11.90 (7.23)	54.13 (23.10)	31.60 (23.64)	1.58 (2.84)	0.00	0.00	0.00
	5	17.15 (9.82)	5.31 (3.30)	20.53 (11.57)	41.78 (23.62)	39.13 (21.88)	5.16 (3.12)	19.13 (42.77)	0.08 (0.18)	0.00
	6	2.29 (2.12)	1.57 (0.98)	0.00	45.85 (26.10)	52.39 (29.56)	145.49 (81.75)	0.58 (1.30)	11.99 (26.81)	1.47 (3.29)
	7	0.00	0.00	0.00	0.00	0.00	0.01 (0.01)	74.13 (41.50)	0.48 (0.51)	30.15 (67.42)
	8	16.05 (35.88)	0.82 (1.84)	0.00	0.00	0.00	0.00	2.76 (1.70)	45.51 (25.47)	5.96 (3.35)
	9	0.00	0.00	2.13 (4.76)	13.59 (30.39)	2.89 (6.46)	0.23 (0.52)	0.00	0.27 (0.17)	109.85 (61.47)

Table A3.2 Mean number of initial 2000 particles within each patch after 6 days (Patch and Source: 1= Peveril, 2=Swanage, 3= Brownsea, 4=Poole Port, 5=Castle Pier, 6=Sandbanks, 7=Boscombe, 8=Hengistbury, 9=Long Groyne).

		Destination								
		1	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**
Source	**1**	0.25 (0.09)	0.09 (0.04)	0.06 (0.12)	0.37 (0.12)	0.21 (0.05)	0.04 (0.03)	0.00	0.00	0.00
	2	1.64 (0.54)	0.17 (0.05)	0.18 (0.10)	0.90 (0.32)	0.55 (0.11)	0.10 (0.06)	0.00	0.00	0.00
	3	11.43 (0.91)	3.66 (0.80)	4.88 (1.08)	22.96 (1.03)	13.27 (0.88)	1.89 (0.48)	0.00	0.00	0.00
	4	6.32 (1.01)	2.42 (0.50)	7.55 (1.47)	32.30 (1.31)	12.50 (0.68)	2.12 (0.46)	0.00	0.00	0.00
	5	5.96 (0.44)	2.07 (0.70)	4.98 (0.41)	14.90 (0.68)	8.07 (0.35)	0.99 (0.24)	0.00	0.00	0.00
	6	5.27 (0.44)	1.25 (0.13)	4.13 (0.72)	12.46 (0.93)	6.04 (0.37)	27.39 (7.17)	0.00	0.00	0.00
	7	10.47 (0.91)	3.41 (0.51)	2.21 (0.68)	12.41 (1.11)	9.80 (2.39)	2.53 (0.96)	0.05 (0.05)	0.63 (0.18)	0.01 (0.01)
	8	2.75 (1.08)	1.58 (0.92)	0.22 (0.06)	1.23 (0.05)	1.82 (0.25)	1.06 (0.22)	1.31 (0.60)	15.41 (1.36)	0.32 (0.13)
	9	0.14 (0.16)	0.22 (0.20)	0.00	0.00	0.07 (0.02)	0.07 (0.04)	0.41 (0.19)	6.76 (0.53)	0.17 (0.13)

Table A3.3 Mean number of initial 2000 particles within each patch after 14 days (Patch and Source: 1= Peveril, 2=Swanage, 3= Brownsea, 4=Poole Port, 5=Castle Pier, 6=Sandbanks, 7=Boscombe, 8=Hengistbury, 9=Long Groyne).

		Destination								
		1	2	3	4	5	6	7	8	9
Source	**1**	0.01 (0.01)	0.00	0.04 (0.04)	0.32 (0.22)	0.07 (0.04)	0.00	0.00	0.00	0.00
	2	0.09 (0.09)	0.02 (0.02)	0.10 (0.07)	0.35 (0.24)	0.19 (0.05)	0.01 (0.01)	0.00	0.00	0.00
	3	0.49 (0.09)	0.79 (0.23)	2.62 (0.31)	6.93 (1.32)	4.02 (0.26)	0.41 (0.12)	0.00	0.00	0.00
	4	0.29 (0.06)	1.00 (0.19)	2.48 (0.54)	10.74 (1.31)	5.27 (0.60)	0.49 (0.09)	0.00	0.00	0.00
	5	0.17 (0.11)	0.63 (0.27)	1.42 (0.31)	5.60 (0.78)	2.99 (0.41)	0.20 (0.07)	0.00	0.00	0.00
	6	0.29 (0.15)	0.35 (0.15)	1.21 (0.30)	4.74 (1.35)	2.23 (0.44)	27.80 (5.76)	0.00	0.00	0.00
	7	0.31 (0.07)	0.72 (0.22)	1.28 (0.37)	5.02 (0.45)	2.90 (0.40)	0.39 (0.10)	0.27 (0.07)	0.41 (0.05)	0.00
	8	1.36 (0.35)	1.85 (0.18)	0.62 (0.20)	3.02 (1.06)	3.45 (0.45)	1.08 (0.16)	7.18 (0.70)	8.52 (1.00)	0.01
	9	0.90 (0.15)	1.14 (0.23)	0.19 (0.10)	1.39 (0.40)	1.66 (0.21)	0.56 (0.10)	4.17 (0.91)	4.98 (0.53)	0.03 (0.06)

Table A3.4 Mean number of initial 2000 particles within each patch after 30 days (Patch and Source: 1= Peveril, 2=Swanage, 3= Brownsea, 4=Poole Port, 5=Castle Pier, 6=Sandbanks, 7=Boscombe, 8=Hengistbury, 9=Long Groyne).

		Destination								
		1	2	3	4	5	6	7	8	9
Source	**1**	0.01 (0.02)	0.00	0.00	0.01 (0.01)	0.00	0.00	0.00	0.00	0.00
	2	0.00	0.01 (0.01)	0.02 (0.02)	0.09 (0.06)	0.03 (0.03)	0.00	0.00	0.00	0.00
	3	0.19 (0.09)	0.10 (0.02)	0.18 (0.02)	1.18 (0.24)	0.33 (0.07)	0.04 (0.01)	0.00	0.00	0.00
	4	0.35 (0.18)	0.29 (0.10)	0.69 (0.28)	3.93 (1.51)	0.68 (0.28)	6.18 (13.32)	0.00	0.00	0.00
	5	0.25 (0.10)	0.16 (0.12)	0.32 (0.36)	1.96 (1.86)	0.45 (0.27)	0.03 (0.02)	0.00	0.00	0.00
	6	0.15 (0.04)	0.12 (0.04)	0.20 (0.06)	1.27 (0.19)	0.21 (0.03)	15.42 (10.66)	0.00	0.00	0.00
	7	0.27 (0.04)	0.11 0.02)	0.18 (0.04)	1.20 (0.14)	0.35 (0.04)	0.07 (0.05)	0.14 (0.10)	0.03 (0.05)	0.00
	8	0.59 (0.10)	0.38 (0.11)	0.51 (0.12)	2.74 (0.30)	1.49 (0.19)	0.74 (0.08)	1.14 (0.21)	0.14 (0.03)	0.00
	9	0.39 (0.08)	0.22 (0.03)	0.14 (0.09)	1.42 (0.22)	0.73 (0.11)	0.52 (0.06)	0.47 (0.14)	0.04 (0.05)	0.01 (0.01)

Table A 3.5 Results of timed search, SACFOR abundance of each species present at each of the nine modelled sites plus existing data from the Boscombe Artificial Surf Reef (BASR).

	Species			
Site	*Styela clava*	*Gibbula umbilicalis*	*Patella depressa*	*Semibalanus balanoides*
Peveril Point		C	A	C
Swanage		O	C	A
Brownsea (Pottery) Pier	R	F		A
Poole Port	O	P		A
Castle Pier	R	O		A
Sandbanks		O	R	A
Boscombe			R	A
Hengistbury				O
Long Groyne		O	R	C
BASR	A			