The ecology of marine tidal race environments and the impact of tidal energy development

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Abstract

Marine tidal race environments undergo extreme hydrodynamic regimes and are favoured locations for offshore marine renewable tidal energy developments. Few ecological studies have been conducted within these complex environments, and therefore, ecological impacts from tidal energy developments remain unknown. This thesis aimed to investigate the ecological aspects of marine tidal race environments in two themes, using a combination of field-based sampling techniques.

I first examined the natural ecological variation of a marine tidal race environment at the spatial and temporal scale. These studies were based on the benthic and intertidal communities within the Alderney Race tidal environment, Alderney. My results suggest that both communities vary in species diversity and composition, at different spatial gradients and timescales. Species showed opportunistic or resilient life history characteristics, highlighting the overall influence of the strong hydrodynamic conditions present.

I then explored the ecology of a marine tidal race environment within a renewable tidal energy development site. These studies were based within the European Marine Energy Centre's tidal energy development site, Orkney. Here, I investigated ecological variation in terms of fish interaction and benthic assemblage structure with a deployed tidal energy device, and, the structure of intertidal communities within the overall development site.

Interestingly, my results indicated species-specific interactions with the deployed tidal energy device, which was related to species' refuge or feeding behaviour. These results also imply that different communities show varied spatial and temporal heterogeneity within a development site, associated with the complex interplay of abiotic and biotic processes.

This work begins to reveal the ecological consequences of tidal energy development, with single devices acting as potential short-term artificial reef structures. Further research is recommended within these environments, with reference to how the hydrodynamic regimes directly influence these communities, and, the overall ecological consequences of future large-scale tidal energy development scenarios.

Declaration

All work presented in this thesis is my own, with the following acknowledgements:

Most of the data have been collected by me, where collected with others, or datasets compiled by others, I cite appropriately throughout. This includes all reference published software and programming code.

All chapters have received editorial advice and comment from my thesis supervisors David Orme (Imperial College London) and Mike Eggleton (ARE Ltd).

Chapter 4 is co-authored with the operations manager of OpenHydro Ltd, Sue Barr and David Orme. This work was a collaborative project with OpenHydro Ltd, which included the development of the sampling method and data collection.

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"The sea, once it casts its spell, holds one in its net of wonder forever."

Jacques Yves Cousteau

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Chapter 1. Introduction

1.1. The marine tidal race environment

The total marine landscape comprises of a diversity of environments, ranging from the deep abyssal plains to the intertidal zones. In this thesis I focus on marine tidal races, which are an uncommon and little studied marine environment type. The environment is specifically found in marine coastal regions that experience extremely strong tidal current velocity flows (> 3 knots; > 1.5 metres per second (m/s)), such as straits, channels, narrows and offshore locations (Connor et al. 2004). Races extend across several water depth zones, such as the benthic circalittoral (> 30 metres), shallow sub-littoral (10 – 20 metres) and intertidal shoreline zones (Brown and Collier 2008). Therefore this environment has been given a range of descriptions and titles including tidal rapids, fast tidal-swept systems, and very strong tidal stream locations (Hiscock 1996; Connor et al. 2004).

In general, marine tidal currents are diffuse across the marine landscape, driven by the gravitational effects of the planetary motion of the sun, moon and earth (Bahaj 2011). The extreme tidal currents observed within marine tidal race environments are the added result of physical and topographic coastline constrictions. It occurs where the tidal flow is forced between islets, islands or coastlines, accelerating the present tidal currents (for example, Figure 1.1; Shields et al. 2009). The nature of these intense hydrodynamic conditions is also heavily dependent upon the present physical and geological characteristics, such as bathymetric roughness (seabed topography), wave exposure and proximity to the coastline (Hiscock 1983; Egbert and Ray 2000; O' Rourke et al. 2010; Bailly du Bois et al. 2012).

Due to the potential range of physical properties, zones and geographical locations, individual marine tidal race environments are therefore often regarded as site specific. This enhances their overall rarity and uncommonness status across the global marine landscape, as a whole (Hiscock 1996; Egbert and Ray 2000; Connor et al. 2004). Despite this, a number of marine tidal race environments are found within and surrounding the territorial waters of the United Kingdom (UK) (Figure 1.2; Bahaj 2011). This is based on computerised tidal current resource hydrodynamic models, which predict the mean tidal flow, tidal range and annual tidal power estimates throughout this geographic region (BERRa 2008). Locations identified as potential marine tidal race environments from these resource assessments include the

Scottish Isles, Wales, Isle of Man, South East and the Channel Islands. For this study, I selected the North-East Scottish Isles and the Channel Islands as study locations.

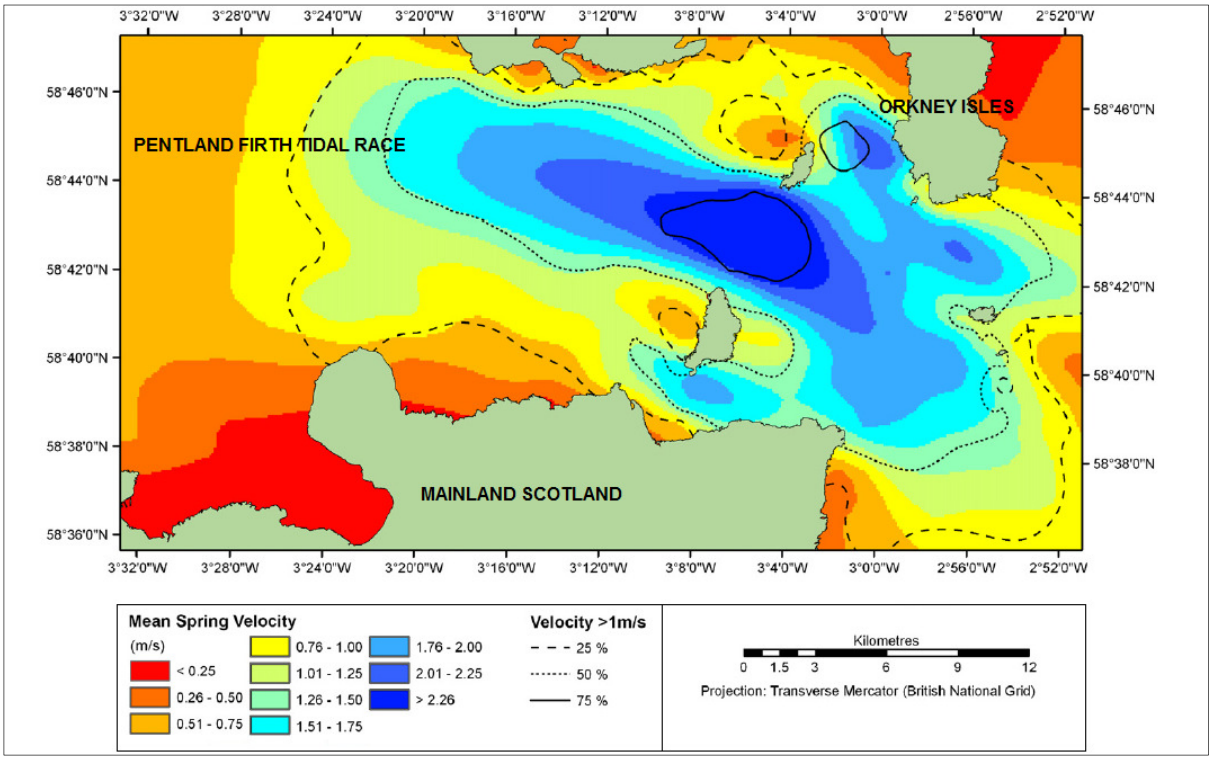


Figure 1.1. Example of a marine tidal race environment, the Pentland Firth and the corresponding tidal current regime. This marine tidal race environment is located between Scotland and the Orkney Isles, with the tidal current regime based on the mean spring velocity (ms^{-1}) and locations where velocity is > $1ms^{-1}$ for 25%, 50% and 75% of the time (adapted from Shields et al. 2009).

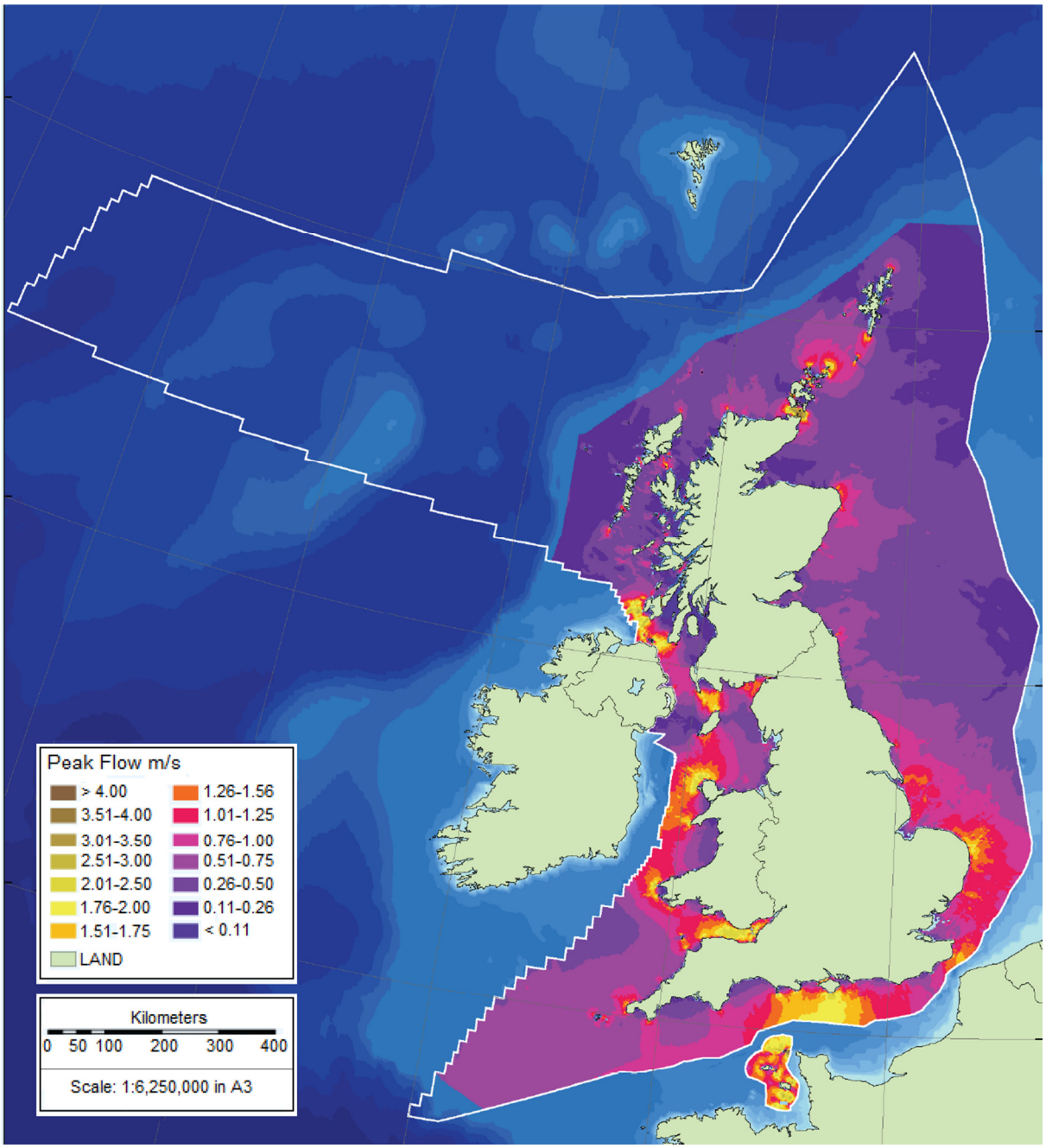


Figure 1.2. Mean tidal current spring peak flow (metres per second (m/s)) within the UK continental shelf and Channel Islands territorial waters. Extent of territorial waters represented by the white line (taken from BERR 2008b). Orange and brown colours indicate flow speeds characteristic of tidal races. The peak flow atlas map is based on the Proudman Oceanographic Laboratory (POL) High Resolution Continental Shelf Model (HRCSM). The model has a resolution of 1/60° latitude by 1/40° longitude and a horizontal resolution of 1 nautical mile (1.8 km).

1.2. The ecology within marine tidal race environments

Marine tidal race environments support different types of ecological communities, due to the varying geological characteristics, extent of water depths and regional, geographical locations. Past species and habitat records show that communities within marine tidal race environments comprise of benthic sponge communities, sub-littoral crustacean and fish assemblages, intertidal macroalgae and their related faunal species (Connor et al. 2004). These communities generally comprise of few species, primarily a result of the dynamic nature of the hydrodynamic regimes and associated processes, and often described as barren or species poor systems (Hiscock 1983; Shields et al. 2011).

Despite the range and influence of other abiotic mechanisms (water temperature and light availability) and biotic processes (competition and predation), including their complex, combined interactions (water depth, light penetration and wave action), water flow (including flow intensity and frequency) are regarded as one of the most important determinants of the marine ecological community structures (Hiscock 1983; Connor et al. 2004; Shields et al. 2011). Numerous past studies show that the movement of water, in the form of tides, currents or waves can heavily influence the larval establishment, growth, feeding, behaviour, reproduction and survival of species (Hiscock 1983; Burrows et al. 2008). Abiotic mechanisms, such as water movement, can therefore determine species presence and hence shape assemblage structure and community distribution (Kostylev et al. 2001; Balata et al. 2006; Denny 2006). Enhanced water movements which occur frequently within marine tidal race environments, are also known to cause organism dislodgement, leading to removal or mortality, whilst reducing the potential for important biotic interactions, such as reproduction (Hiscock 1983; Shields et al. 2011). As such, marine tidal race environments are represented by species and communities that exhibit distinct life history strategies that enable them to cope with the extreme hydrodynamic conditions and associated disturbance regimes from increased scouring and storm events (Hiscock 1983; Sousa 1979; Shields et al. 2011). For example, within the benthic zone of these environments, species and habitat records frequently describe mixed hydroid or bryozoan assemblage compositions, and the presence of cnidarian and crustacean species (Connor et al. 2004). These communities primarily display faunal turf suspension feeding structures or opportunistic mobile traits, which are related to the variable availability of food from the water column and seafloor (Hiscock 1983; Okamura and Partridge 1999). These types of traits are also represented at the intertidal shoreline zone within these environments, which comprise of tidal swept fucoid and kelp macroalgal

habitats, mussel or barnacle mosaics and opportunistic coloniser macroalgae species (Sousa 1984; Brown and Swearingen 1998; Nishihara and Terada 2010). In particular, macroalgae located within these environments are known to have larger and thicker frond densities, adapted to endure the increased mechanical stress from the intense hydrodynamic conditions (Pratt and Johnson 2002; Stevens et al. 2002; Wernberg and Connell 2008). These ecological descriptions and patterns (morphological forms) have also been found within marine environments which undergo other forms of extreme environmental and hydrodynamic conditions, such as intense wave exposed marine environments (Denny 2006; Burrows et al. 2008).

Although the general ecological characteristics of tidal race communities are known, Hiscock (1983) reported that more detailed research is required; to reveal the ecology of such communities and their complex relationship to other hydrodynamic conditions, environmental processes and biological factors. This has been followed by studies (Underwood et al. 2000; Menge et al. 2005) which suggest that such research should also consider the ecological variation across spatial and temporal gradients, to fully understand such hydrodynamic systems over space and time. Two decades on, Shields et al (2011), found that such gaps in knowledge still exist, particularly with respect to defining and characterising the overall ecology of marine tidal race environments. This is due to the associated environmental conditions (extreme hydrodynamics and weather conditions) within these environments restricting ecological sampling efforts, method design and research activities in the past (Shields et al. 2011). Indeed, these environments are still given basic generic descriptions both at the environmental and ecological level, with descriptions such as heavily exposed, physically energetic environments or tidal swept community types (Connor et al. 2004). Other research studies also now exist which define the ecology within comparable marine environments (for example, extremely wave exposed hydrodynamic environments), which can begin to enhance the current qualitative descriptions of marine tidal race environments (see Connor et al. 2004 and Denny 2006). For example, these comparative studies may aid the preliminary ecological characterisation of marine tidal race environments (in terms of diversity estimates or morphological functional form); help form and test hypotheses associated with ecological patterns (spatial and temporal patterns); or develop suitable surveying methods and techniques to sample with these extreme environments. Despite these similar studies, Hiscock (1983) and Shields et al (2011), still primarily suggest the development of in-depth study of environments which undergo strong

tidal flow conditions, due to the differences in the physical properties of tides, currents and waves (wave vs. tidal flow intensity and frequency), subsequent ecological influences and their complex abiotic/biotic interactions.

Such knowledge of these unique environments is inherently important, not only to fully define the ecology within them, but to also further our understanding of fundamental ecological principles. For example, these environments provide useful examples to further our understanding of the roles of disturbance, succession and life history traits (Tilman 1990; Steneck and Dethier 1994; Menge et al. 2005). Therefore this thesis describes the ecology of the benthic (Chapter 2) and intertidal (Chapter 3) regions within the marine tidal race environment, in terms of species biodiversity, assemblage composition and life history descriptions, over space and time, using current research strategies and adaptable sampling method designs. I assess these ecological patterns within the marine tidal race environment, the Alderney Race, located within the Channel Islands. Finally, such research is timely because of the increasing interest in exploiting tidal race systems for energy generation.

1.3. Marine renewable energy development

The demand for energy is increasing dramatically on a global scale. Rising human population, urbanisation and modernisation have led to substantial growth in the energy sector, with predicted global estimates of 57% developmental growth from 2002 – 2025 (Asif and Muneer 2007). Combined with dwindling fossil fuel stocks, concerns for securing viable energy supplies and their associated climate change impacts have driven the energy sector to seek alternative sources of energy (Allan et al. 2008). This includes natural, renewable sources, such as wind, solar, biomass and geothermal energy resources (Angelis-dimakis et al. 2011). Renewable energy sources such as these are regarded as ideal replacements for fossil fuels and can provide long-term sustainable energy supplies that produce significantly reduced carbon dioxide (CO_2) emissions compared to fossil fuels (Asif and Muneer 2007). These energy sources can also enhance the diversity of the global energy supply market and are financially attractive to developing countries such as India and China for economic growth (Allan et al. 2008). In addition, a number of countries have also introduced binding legislation and financial incentives to encourage the use and development of renewable energy sources (Bahaj 2011). For example, the European Commission (EC) recently implemented the Renewable Energy Policy, committing the European Union to reach

renewable energy targets of 20% of their total energy production by 2020 (Directive 2009/28/EC; Bahaj 2011). As a result, the scope for renewable energy sources has started to expand into a number of natural systems, including the marine environment.

Since the last decade, substantial development has been focused at offshore marine energy exploitation, particularly in offshore wind turbine farms. The United Kingdom has established a number of large-scale offshore wind farms to try to meet the government's own renewable energy targets of 15 % by 2020 (Bahaj 2011; Toke 2011) and a total capacity of 1200 megawatts (MW) was reached in 2010. Interest is now however turning to other potential sources of energy within the offshore marine environment, ranging from salinity and thermal gradients to waves and tidal currents (Allan et al. 2008). Tidal current energy generation is a highly lucrative sector that exploits the kinetic energy associated with the predictable tidal patterns and current flows found in the marine environment (Kelly 2007) using suitable energy generators (Couch and Bryden 2006; Bahaj 2011). The global potential for such energy extraction is extremely large, with tidal dissipation extraction across the world continental shelf estimated at 2.5 terawatts (TW) (Egbert and Ray 2000; Bahaj 2011). Tidal energy developments are however primarily selected within offshore sites which exhibit increased velocity flows such as marine tidal race environments, which can potentially generate extremely large, viable quantities of energy (Couch and Bryden 2006; Block 2008). Marine tidal race environments, have therefore received great interest from the tidal energy sector, with potential development sites found within the Isle of Wight, Orkney Isles and Shetland Isles, and also the Channel Islands, with the Alderney Race predicted to produce a total annual yield of 1.343 terawatt-hours (TWh) (Bahaj and Myers 2004; Myers and Bahaj 2005). Overall, the scope for such development within UK territorial waters is extremely large, with a total of 10% of the estimated global tidal energy resource falling within this region (DECC 2009; Bahaj 2011; Matthews et al. 2012).

Currently there are a number of different tidal energy generator device designs available to harness such energy (approximately 200 concept designs), with generator designs following either the vertical axis design or horizontal axis design (O' Rourke et al. 2010; Matthews et al. 2012). Vertical axis device design comprises of generator turbine blades rotating on a vertical axis, which is perpendicular to the direction of the flow of water, such as the Gorlov Helical Turbine (Figure 1.3a). The horizontal device design comprises of a turbine blade rotating on a horizontal axis, which is parallel to the direction of the flow of water, such as the OpenHydro Ltd open turbine (Figure 1.3b).

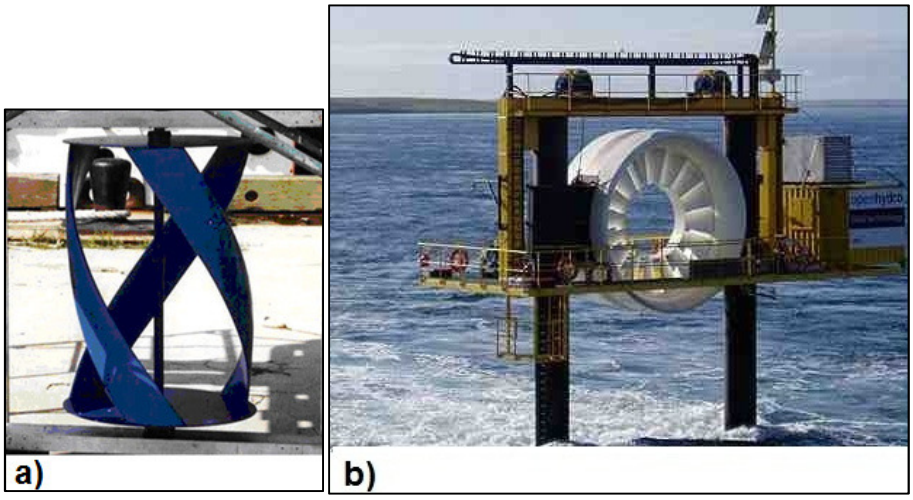


Figure 1.3. Tidal energy generator device examples. Vertical axis Gorlov Helical Turbine a) and the OpenHydro Ltd horizontal axis design b), (taken from O' Rourke et al. 2010; OpenHydro Ltd 2012).

The focus of the tidal energy sector is, at present, centred round the development of single test or commercial generator devices and their associated activities, specifically within offshore tidal energy test sites. The production in volume of commercially viable tidal energy devices and development schemes is still at an early research and design phase for many key developers (Bahaj 2011). Such technology is still extremely new and often regarded as pioneering. As a result, prototypes are still undergoing fundamental engineering assessments and are subject to a number of Technology Readiness Level (TRL) protocols (Bahaj 2011). Turbine device concept designs are first modelled in laboratory simulated conditions, then tested at 1:100 scale in tanks, followed by 1:10 scale devices built and transferred to offshore tidal energy test sites for extensive field tests. Test devices then undergo further rigorous design alterations and research within test sites, before commercial device manufacture (O' Rourke et al. 2010). Offshore tidal test sites, such as the European Marine Energy Centre (EMEC), based in the Orkney Isles, therefore play an important part in the preliminary designs for full commercial scale tidal energy development. These sites also permit associated activities such as pile-driving and sub-sea cable attachment exercises to be tested in intense hydrodynamic conditions (Norris and Droniou 2007).

For full scale commercial tidal energy development, the device type, size and number are site specific. This is based on the water depth, bathymetry and tidal current flow levels at the site.

Sites also require detailed research and planning of device deployment locations, energy resource assessments and operational activities in order to exploit energy from the tidal currents efficiently (O' Rourke et al. 2010). As such, few actual commercial developments currently exist, with development plans within selected areas still to be finalised. The future scope and potential for tidal energy development is however rapidly increasing, with a number of schemes proposed throughout the UK in the next five years. These range from the small scale (1 – 10 device deployments in one site) such as the Orkney Isles and Ramsey Sound (Figure 1.4a) to the large scale array developments (100 - 250 device deployments in one site; Figure 1.4b) such as the Alderney Race (Block 2008; Shields et al. 2011). Overall, due to the potential scale of these developments and encouragement from the political and energy sectors, this type of new anthropogenic activity could soon be found throughout the marine tidal race environment landscape.

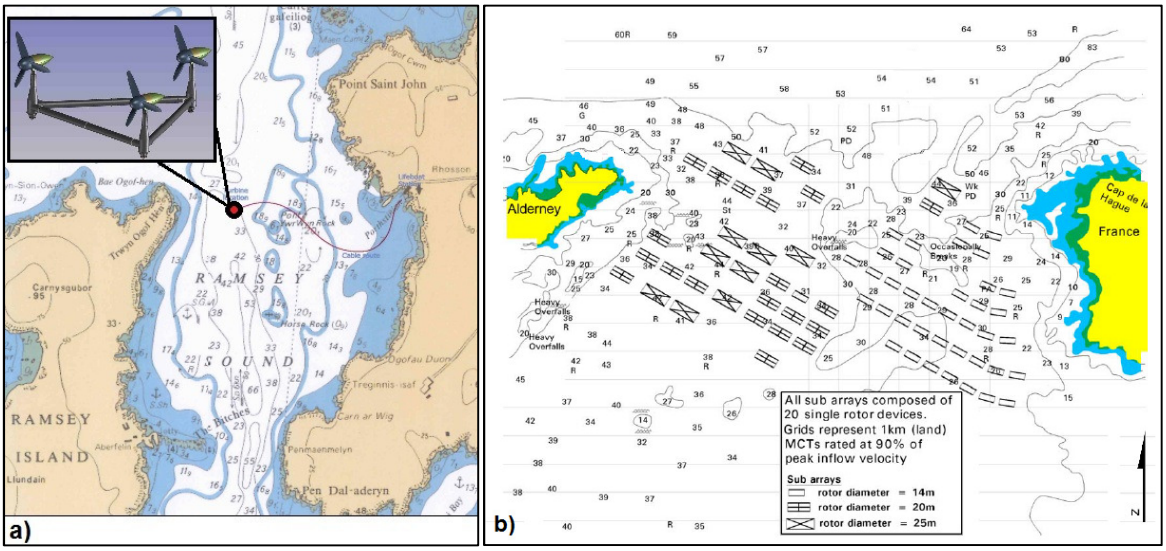


Figure 1.4. Examples of a proposed single, small-scale tidal energy device development in the Ramsey Sound, Wales a) and a simulated model of a large-scale tidal energy device array in the Alderney Race, Channel Islands b). a) represents the Deltastream tidal energy device, with the specific selected location within the Ramsey Sound represented as the red circle, and connecting sub-sea cable represented by the red line. b) represents a simulated layout of tidal energy devices (Marine Current Turbine devices) based on the offshore site depths, size and length with different rotor diameters shown (adapted from Myers and Bahaj 2005; Tidal Energy Ltd 2009).

1.4. Detecting anthropogenic impact in the marine environment

The wide-scale anthropogenic use of the marine environment is ever increasing. From transportation and fisheries, to coastal protection and waste disposal, the marine environment provides a wealth of ecosystem functions and services (Eastwood et al. 2007). The growing intensity, scale and range of anthropogenic activities have increased concerns relating to environmental decline and ecosystem degradation (Inger et al. 2009). Consequently, this has fostered numerous studies into describing the environmental impact from such anthropogenic activities in the marine environment, in an attempt to identify the change of the natural state of the environment and reduce the potential for environmental decline.

A large number of these studies are devised for Environment Impact Assessments (EIA) or Strategic Environmental Assessments (SEA), instigated by commercial bodies for development planning and governmental consents. Such assessments aim to qualitatively evaluate and mitigate for the environmental impacts associated with new or on-going anthropogenic activities (Terlizzi et al. 2005; Langhamer 2010). These follow environmental impact conceptual framework methods, to describe the level, scale and effect of the impact specific to the type of anthropogenic use within a chosen location or environment (Gordon and Longhurst 1979). The framework evaluates: the source and nature of the specific modification, the affected components, the consequence of change, and the component availability and status within the total natural resource.

To define and measure the scale of environmental impact and change, these assessments frequently treat ecological aspects and parameters of the environment as the affected components. This can include organism presence, species biodiversity descriptors, assemblage composition, habitat integrity and ecosystem functionality status (Terlizzi et al. 2005; Airoldi et al. 2008; Wilhelmsson and Malm 2008). These ecological aspects have revealed that the marine environment can undergo a variety of different impacts from anthropogenic activities such as fishing, dredging, and pollution (Eastwood et al. 2007). Marine environmental impacts are commonly documented as: physical loss (smothering, obstruction), non-physical disturbance (noise, visual), damage (material extraction, abrasion), toxic (compounds) and non-toxic concentrations (nutrient concentrations) impacts (Eastwood et al. 2007). The consequences of such impacts can cause direct ecological effects, in terms of organism tissue and endocrine damage, mortality, and habitat loss, or, indirect effects of;

alteration of species interaction or behavioural response, trophic level interactions, larval settlement disruption or reproduction effort disturbance (Gill 2005; Eastwood et al. 2007).

Academic and scientific research has significantly contributed to this current level of environmental impact knowledge, by advancing ecological monitoring techniques and impact analyses, such as the development of the Before After Control Impact method (BACI) or the After Control Impact method (ACI) (Terlizzi et al. 2005). These methods aim to identify anthropogenic impacts and the degree of environmental change, by quantitatively measuring ecological aspects either before and/or after the start of an anthropogenic activity (Langhamer 2010). A number of studies have since expanded this technique to also assess these at varying spatial and temporal scales through the inclusion of additional control sites and on-going monitoring (Underwood 1992). These consider the natural ecological variation and their potential varying patterns of response (direct and indirect) to anthropogenic impact over space and time (Langhamer 2010). This can also include assessing the interaction between both ecological and key physical processes, to determine the overall impact in the marine environment (Terlizzi et al. 2005). Such methods provide extremely useful in-depth knowledge on the process of environmental change, and could therefore be applied to assess impacts from new anthropogenic activities, such as the development of tidal energy.

Few in-depth environmental impact assessments have been undertaken so far for the tidal energy sector, due to limited current technology information and lack of development plans (Inger et al. 2009; Frid et al. 2012). In addition, potential important data sources or studies for UK assessments may also be publically inaccessible, due their commercial sensitivities (House of Commons 2013). In the past, publically available tidal energy development EIA's and research have had to use alternative sources of information from other comparable anthropogenic activities to evaluate potential environmental impact. This has included sources from offshore marine renewable energy developments (wave, wind and tidal barrage energy systems), marine energy production (oil and gas exploitation) and coastal development schemes and activities (benthic zone seismic surveys, moorings, breakwaters developments). These activities are primarily associated with increased vessel use, deployment of structures, energy resource extraction and production, which occur during the total life cycle of these anthropogenic activities (i.e. installation, operation and decommissioning). The environmental impacts associated with these types of activities are primarily acknowledged to affect the pelagic, benthic and intertidal systems, where such activities largely occur.

These impact studies show that such activities can both positively and negatively affect the natural process of primary productivity, species biodiversity, assemblage composition and habitat structure (Page et al. 1999; Gill and Taylor 2001; Gill 2005; Petersen and Malm 2006; Wilhelmsson and Malm 2008; Kirby 2010). The associated negative impacts are described as physical loss or displacement, either during the initial deployments of these man-made structures (pile-driving monopole and general structure deployments) or operational activities (turbine noise generation, electromagnetic field alterations (EMF) from sub-sea cables) and can also encourage bio-fouling or invasive species growth (Petersen and Malm 2006; Wilhelmsson and Malm 2008; Langhamer and Wilhelmsson 2009; Langhamer et al. 2009; Maar et al. 2009; Langhamer et al. 2010). These suggest that the overall integrity of the ecosystem may therefore decline, due to such activities disrupting and altering the present natural assemblage or habitat structure, allowing the ecosystem to become more susceptible to the presence of fouling, non-native or different species compositions. Associated positive impacts are described as increasing species biodiversity and habitat integrity, which is also affiliated to the deployment and permanency of these structures on the seafloor. The addition of new structures to the marine environment can enhance organic material production from colonising species, create artificial reef scenarios (including fish aggregation devices (FAD's)), increase habitat complexity, and prevent other more damaging impacts, such as dredging, due to large spatial requirements for such energy exploitation developments (> 1 km^2) (Inger et al. 2009; Langhamer and Wilhelmsson 2009; Hiscock et al. 2010; Langhamer 2010; Langhamer et al. 2010).

Overall, using information from these similar types of offshore anthropogenic activities has been initially useful to outline potential environmental impacts from tidal energy development activities. True patterns of environmental impact may have however, been missed, due to differences related to the type of device technology and their action within the marine environment. For example, recent studies have begun to outline the potential influence of tidal energy devices upon both the local (< 1 km) and regional (1-10 km and > 10 km) hydrodynamic regimes and coastal processes within the marine environment (Neill et al. 2009; Kirby 2010; Kadiri et al. 2012; Neill et al. 2012). A number of past studies show that the deployment of new structures to the marine environment can influence the hydrodynamic processes surrounding the structure and the local environment (Gordon and Longhurst 1979; Retiere 1994; Wilhelmsson and Malm 2008; Langhamer et al. 2009; Langhamer 2010). Shields et al (2011) describe that these deployments will alter the

hydrodynamic regimes present, in terms of: decelerate flow rates upstream and downstream of the deployed structure, accelerate flows around the structure and also create a turbulent wake. These studies also suggest that tidal energy developments could alter hydrodynamic regimes across the local and regional spatial scales, through the tidal energy extraction properties from individual devices themselves, to the potentially large regional (spatial) extent of the device array developments (Shields et al. 2011; Ahmadian et al. 2012).

As such, this may cause unpredictable or unknown environmental impacts, which may not have been included in previous impact assessments or other marine energy research studies (Langhamer 2010; Neill et al. 2012). Tidal energy technology and their associated activities therefore need extensive in-depth ecological considerations to assess potential environmental impacts (Gill 2005). In addition, due to the unpredictability and potential scale of such impacts throughout their life cycles, these considerations should also be extended when selecting the ecological aspects, for effective impact measurements. Martin et al (2005) suggest that certain ecological aspects (for example specific species) may not show measureable responses and therefore misinterpret the pattern of environmental change. Therefore, a wide-range of appropriate ecological aspects and parameters should be chosen, which are directly affiliated with the selected location or zone and the type of anthropogenic activity.

Overall, to assess potential environmental impacts associated with tidal energy development activities, in-depth studies should be developed that consider the type or scale of technological use, the natural variability of the natural resource and the pattern of environmental change, which use appropriate ecological aspects (Gill 2005; Langhamer et al. 2010). My research consequently uses this approach, by implementing a range of ecological studies over different spatial marine regions and temporal scales. This includes the benthic and intertidal regions, with ecological aspects including species abundance, biodiversity, taxonomic or functional form descriptors, assemblage composition, and habitat types. Selected temporal scales follows key marine ecological research, which denote important ecological time-scales such as seasonal and annual time periods for both regions (Underwood et al. 2000; Menge et al. 2005; Balata et al. 2006).

These can give a good ecological perspective and measurement of environmental impacts from anthropogenic activities in the marine environment, which have been used in a variety of impact studies (Terlizzi et al. 2005; Langhamer 2010). The level of anthropogenic activity

is centred round the current status of the tidal energy sector. This is primarily focused at the impact at the operational stage of a single test tidal energy device and the overall impact of a designated tidal energy test site. These were conducted surrounding the OpenHydro Ltd test device, within the EMEC Fall of Warness marine tidal race environment. I present research on pelagic effects of tidal energy generation, using adaptable sonar and video data sampling method designs and techniques to examine fish behavioural responses with the deployed tidal energy device (chapter 4). I also present research on the benthic assemblages associated with the deployed tidal energy device (chapter 5) and the intertidal community compositions found on coastlines adjacent to the tidal energy development site (chapter 6).

1.5. Rationale for this study

Overall, limited in-depth ecological information exists on marine tidal race environments. This is due to the general difficulties associated with sampling within these extreme marine environments (increased hydrodynamics regimes and poor weather conditions), and their natural rarity across the marine landscape as a whole. Past research comprises of anecdotal species lists, and generalised habitat description records, derived from local or national biological recording schemes, and also from a small number of scientific studies. This also includes preliminary information from other comparable marine environments, such as wave exposed systems. These studies are particularly restricted in terms of scale, first at the spatial scale, with records limited to one biological zone region (i.e. benthic or intertidal region), and second at the temporal scale, with few studies repetitively sampling these environments across different time periods (i.e. seasonal or annual scales).

The need for such information has however, increased dramatically over the last decade, predominately for data gathering exercise requirements during tidal energy development environmental impact assessments. In addition, the ecological importance of this type of marine environment has also increased, based on the potential large scale of these new developments and also their rarity. As such these environments are recognised within the 'Tidal Rapids: UK Biodiversity Action Plan', governmental biodiversity strategy, which was designated in the late 1990's (UKBAP 2008).

Despite these efforts and the added interest from the renewable energy sector, there are still few quantitative studies which have examined the ecological impacts from such anthropogenic developments. This is also the result of limited information regarding tidal energy device designs, operational activities, development scenarios and the overall newness of this type of anthropogenic activity within the marine environment. Coupled with the present lack of baseline ecological knowledge, the ecological impacts from such new anthropogenic activities have therefore been extremely difficult to quantify effectively (Gill 2005; Shields et al. 2009). Therefore, this study aims to begin to understand the ecology of these extreme environments and aid the overall understanding of potential ecological impacts from such developments.

1.6. Overall thesis aims and structure

The overall aim of this thesis is to investigate the ecology of marine tidal race environments and the impact of tidal energy development. Here, I use species abundance, biodiversity, composition, functionality and habitat descriptors to examine the ecological aspects of these environments. This includes exploring these ecological aspects at both the spatial and temporal scale, using applicable and adaptable field-based methods. I specifically focus on the benthic and intertidal regions within this environment, to give a wider perspective of the ecological communities and their interaction to tidal energy developments, as a whole.

The structure of this thesis is therefore divided into two research themes; first, I examine the ecology within a 'natural' marine tidal race environment, namely the Alderney Race, Channel Islands. **Chapter 2** investigates the benthic communities within this system, at the spatial and temporal scale, using a fine-scale approach. **Chapter 3** examines the intertidal regions of this system, to investigate the ecological variation within this type of under-sampled environment. My hypotheses within these two chapters test the ecology within these marine regions are related to the extreme environment type, and that their ecological patterns vary across the different measured spatial and temporal scales.

I then explore the ecology within the EMEC tidal energy development test site, which is located in the Fall of Warness marine tidal race environment, Orkney Isles. For this I aim to examine potential ecological interactions with tidal energy technology at its present state, which in the UK, is currently focused at the operational single test device and test site development stages. In **Chapter 4**, I assess the direct response of fish species to a deployed tidal energy device, using a combination of field-based methods. **Chapter 5** examines the benthic assemblages and associated habitats surrounding a deployed tidal energy device, over time. **Chapter 6** then investigates the intertidal communities found within the overall development site. My overall hypotheses within these three chapters test that the deployed tidal energy test device alters the ecology within these marine regions across the measured spatial and temporal scales. Finally in **Chapter 7**, I summarise my key findings, research caveats and recommendations for future study.

Chapter 2. Fine scale benthic assemblage patterns within a marine tidal race environment, fundamental questions in under-sampled environments?

2.1. Abstract

The benthic regions of the marine environment are often studied using broad-scale sampling methods. In systems which are topographically complex, or poorly defined, a fine-scale approach may be more suitable to determine physical seafloor properties and their associated ecology, with greater accuracy. Here, I use a towed video camera system to determine the physical substrate characteristics and benthic ecology of the Alderney Race. Sampling covers water depths of 30 – 40 metres over six periods during 2009 and 2010.

I show that these uncommon benthic landscapes comprise of a range of substrates, which further define the increased seafloor complexity of this environment. I found that the recorded ecology comprised of a mixture of suspension feeding turf and opportunistic singleton species, which are associated with strong hydrodynamic conditions. Overall, these results show that benthic species assemblage and functional form compositions are similar between the surveyed areas, which alter across the six temporal sampling periods. I also observed that the general physical and ecological relationship was extremely weak. This begins to outline the variety of environmental and ecological processes that can occur within this under-sampled environment. This study provides detailed quantitative description of a little studied marine habitat, which faces the possibility of substantial future impacts arising from the expansion of tidal power generation.

2.2. Introduction

The benthic regions of the marine environment have been a source of interest for ecological, geological, and environmental study for centuries. The last few decades have seen an increasing focus on the quantitative study of the composition of this marine landscape. This has led to a creation of a variety of sampling designs and techniques, to determine seafloor characteristics, their associated species assemblage compositions and wide-scale environmental processes. These designs have included the application of seabed sediment profiling, scuba diving, grab-sampling, towed video camera assessments and commercial fisheries techniques (Parry et al. 2003; Diaz et al. 2004; Bremner et al. 2006). Recently, focus has shifted to a more broad-scale application of these techniques, with the use of new and integrated technologies such as Side Scan Sonar (SSS), Multi-Beam Echosounder Systems (MBES) and Geographical Information Systems (GIS) (Shumchenia and King 2010; Brown et al. 2011; Freitas et al. 2011; Kaskela et al. 2012). These applications are particularly useful to survey large areas (>1 km^2) of the benthic landscape and examine marine species presence, habitat distribution and their association with physical factors, such as seafloor characteristics (Kostylev et al. 2001; Galparsoro et al. 2010; van Rein et al. 2011). Broad-scale recording schemes such as the UK benthic habitat mapping scheme, UKSeaMap and the European Marine Strategy Framework Directive (MSFD), now readily use these techniques for legislative marine biodiversity objectives, eco-system fisheries management practices and the future designation of marine protected areas (MPA) (Connor et al. 2004; MSFD 2008/56/EC; Bosman et al. 2011; Cameron and Askew 2011; Huang et al. 2011; Rice et al. 2012).

The use of broad-scale sampling methods for such studies or schemes may however be inappropriate for certain marine environments, such as offshore marine tidal race environments. Offshore marine tidal race environments are uncommon within the benthic landscape, described as site specific and found across a number of geographical regions (Bahaj 2011). This is due to these environments regarded as being topographically complex due to the presence of differing strong hydrodynamic regimes, geological properties and environmental processes (Bailly du Bois et al. 2012). As a result, little quantitative physical and ecological information or general environmental knowledge (including general regional environmental comparison studies) exist on the benthic regions of these dynamic environments (Gill 2005; Shields et al. 2009). The application of broad-scaled surveys or generalised schemes may miss fundamental environmental processes and ecological interactions within these benthic environments. The majority of these broad-scale schemes

and studies are also frequently conducted at a single temporal sample (van Rein et al. 2011). Benthic physical characteristics and ecology can show natural temporal variation; therefore studies which only include one temporal sampling strategy may miss important temporal patterns (Galparsoro et al. 2010). In addition, a number of the surveying techniques and sampling method designs selected for broad-scale studies may be extremely difficult to use within this type of marine environment. The complex topography coupled with the extreme environmental variables may reduce sampling efficiency; hinder survey objectives or increase surveying costs and project timescales (Shields et al. 2011). Therefore, adopting a more fine-scale, in-depth approach to assess the benthic physical characteristics and ecology in terms of spatial and temporal variation within this environment type, before broad-scale sampling methods may be more suitable (Galparsoro et al. 2010; Gogina et al. 2010). This would be extremely useful to enhance the overall sampling efficiency, practicality and cost effectiveness, particularly for research projects with limited resources or time constraints. For example, a more fine-scale approach could include the comparison of two survey sites within one tidal race environment. Such an approach could first increase the preliminary ecological information of these environments, and then enable more broad-scale studies (such as regional marine tidal race environments ecological comparison assessments) to advance knowledge for climate change, ocean acidification and anthropogenic impacts (Eastwood et al. 2007; Philippart et al. 2011). As these offshore marine tidal race environments are also considered potential as sites for future marine renewable energy exploitation schemes; therefore, increasing quantitative information of this environment would inherently aid future anthropogenic environmental impact assessment studies (Gill 2005; Inger et al. 2009; Shields et al. 2009).

The overall aim of this chapter is to therefore investigate the benthic physical characteristics and ecology of a marine tidal race environment using a fine-scale approach at both spatial and temporal scales. For this study, I selected the marine tidal race environment known as the Alderney Race, found within the Channel Islands. This site has few in-depth physical and ecological records, with the overall study site selected for a future large scale marine renewable tidal energy development (Osiris Projects 2009). I selected the overall spatial extent of this study to fall within the potential development area for tidal energy generation in order to both increase baseline information and to support potential environmental impact assessments of such a future development.

To undertake this study, I surveyed the Alderney Race using a benthic towed camera video sampling technique. Camera video sampling techniques are useful for defining physical characteristics and ecology within complex geomorphological benthic environments such as this (Kostylev et al. 2001; Sánchez et al. 2009). Other sampling techniques such as sediment analysis, grab sampling, or the use of commercial trawls to assess this environment are inappropriate, due to the type of seafloor characteristics recorded (such as pinnacles, bedrock, boulders) in past surveys (BGS 1989; BGS 1990; Admiralty Nautical Charts 2004; Osiris Projects 2009).

This fine-scale study specifically incorporates the spatial (two sites; < 1 km^2) and temporal (seasonal) scale aspects, within a specific water depth band group. Benthic community structures can show distinct changes across different water depth gradients i.e. between 10 – 20 metres and 30 – 40 metres (Post 2008; McArthur et al. 2010). Examining within specific water depth bands can provide greater accuracy of the physical characteristics, ecology and their associated interactions, whilst increasing the primary baseline information within this environment. Here I chose the water depth band between 30 – 40 metres, based on the overall mean water depth of the Alderney Race environment (40 metres), which are also appropriate depths for tidal energy device deployments (Couch and Bryden 2006). The physical characteristics used during this study are defined in terms of physical substrate composition, with the ecology defined as species' abundance or proportion cover, biodiversity, composition and feeding regime type. These are commonly used in a variety of ecological and environmental impact assessments to determine benthic assemblage composition and environmental processes. Here I first test the hypothesis that the benthic physical characteristics and ecology are spatially similar (two sites; < 1 km^2) within the selected water depth band. I then test the hypothesis that these alter across the measured temporal (seasonal) scales. I also test the hypothesis that the physical characteristics and ecology are related to each other and the environment type, overall.

2.3. Methods

Study area

The study area was located within the marine tidal race environment, known as the Alderney Race. This race environment is located between the Channel Island of Alderney and the Cap de la Hague, found on the North-West coastline of France. The race is approximately 4 miles

wide with peak velocity flows from 4.5 m/s^{-1} at -3 hours (relative to high water at Dover) to 4.8 m/s^{-1} at -4 hours (HW Dover), with mean water depths of 40 metres (Bahaj and Myers 2004, Black and Veatch Consulting Ltd 2005). This area is characterised by complex topography, comprised of exposed irregular rocky outcrops, seabed fissures and gullies, with a mixture of gravel type substrates located throughout the island's territorial waters (BGS 1989; BGS 1990; Admiralty Nautical Charts 2004; Osiris Projects 2009; Bailly du Bois et al. 2012).

The study was conducted within two 1 nautical $mile^2$ (nm) squares located within the Alderney Race, known as T60 and T61. These squares represent sites allocated by the local government (Alderney Commission for Renewable Energy, The Alderney States) to the renewable energy developer, Alderney Renewable Energy Ltd (ARE Ltd) for marine renewable tidal energy development. Transect lines were used to survey a smaller area of approximately 0.128 nm^2 within each of the T60 and T61 squares (Figure 2.1). The location of these survey areas was primarily chosen for capturing the target water depths (37- 44 metres), comparable physical properties (substrate, velocity speeds) and to provide sampling efficiency and safety during the study; due to the strong hydrodynamic conditions present (BGS 1989; BGS 1990; Admiralty Nautical Charts 2004; MESH 2008; Neill et al. 2012). These areas were selected based on preliminary data from earlier survey trials completed in 2009, and also bathymetry maps provided by ARE Ltd. Bathymetry information can provide in-depth geological seafloor characteristics and bed-form distributions, which is particularly useful for ecological survey assessments (Brown et al. 2011).

The data presented here were collected across six temporal sampling periods from 2009 - 2010. This included: February 2009, May 2009, August 2009, November 2009, February 2010 and May 2010 respectively. The study was initially designed to sample an additional four more temporal sampling periods, August 2010, November 2010, February 2011 and February 2011. These surveys were cancelled due to severe weather and hydrodynamic conditions.

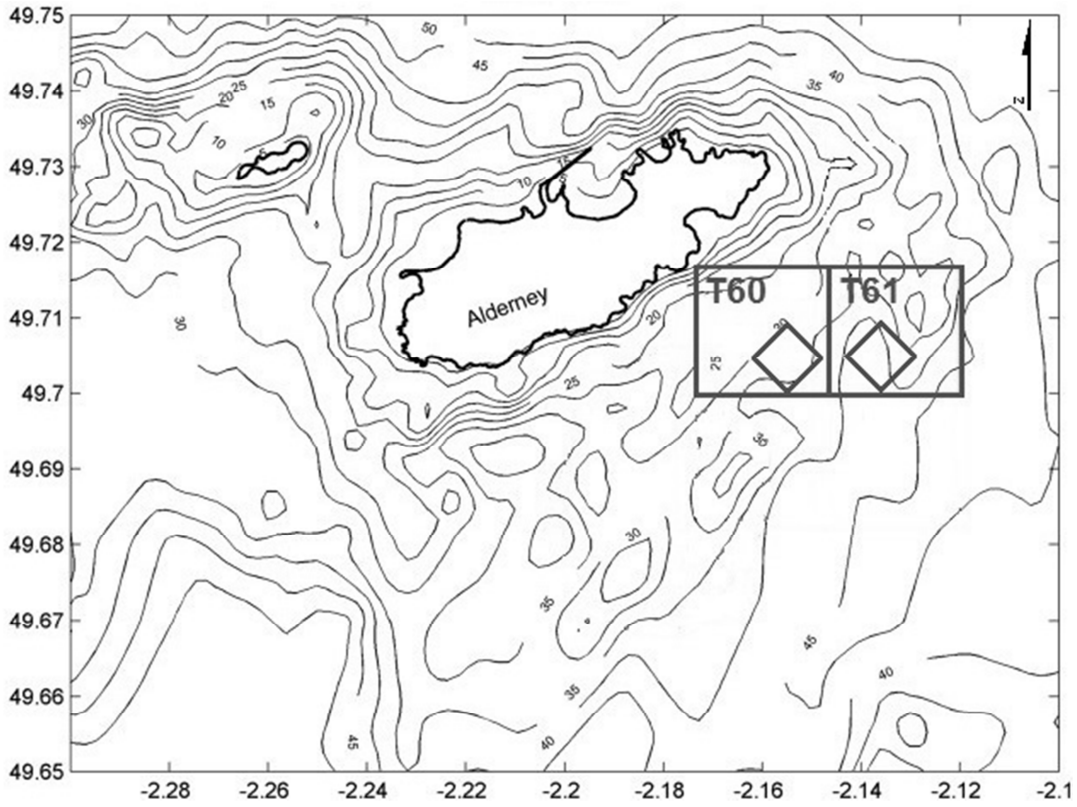


Figure 2.1. Location and bathymetry of the T60 and T61 survey squares within the Alderney Race marine tidal race environment. The T60 and T61 survey squares are represented by the grey squares (area of each square: 1 nm^2), with the smaller survey areas within both squares represented by the grey diamonds (area of each sub-section square: 0.128 nm^2). The survey squares are locations selected for future offshore marine renewable energy development by ARE Ltd. The smaller survey areas are locations selected for the benthic study. Contours represent the water depths (metres), relative to the mean sea level, with overall map coordinates given in decimal degrees (adapted from Neill et al. 2012).

Towed camera video assessment

The benthic physical substrate composition and ecological composition was assessed using towed video camera techniques (following Brown and Collier 2008; Sánchez et al. 2009) within the selected survey areas of the T60 and T61 squares. Using dGPS, five transect lines were located within each survey area, which were approximately 741 metres long (0.4 nm) and approximately 150 metres (0.081 nm) from each other (MESH 2008). The bearing of each transect line was 47°, based on the general tidal flow directions of the Alderney Race, for effective sampling of the seabed in this area (Figure 2.2).

The towed camera video system was deployed at the start of each transect line, with the vessel controlled to drift between approximately 0.4 – 2 knots along each transect line (see Appendix 2 for additional schematic of camera system: Table A.2.1, Figure A.2.1 and dGPS coordinates: Table A.2.2) against the water current direction. The camera was held approximately 1 metre (± 0.5 metres) above the seabed from the vessel by an small A-frame, with the camera (4.3 mm camera lens) facing downwards to continuously video record the seafloor (Coggan et al. 2005; Ierodiaconou et al. 2011). Footage of each transect line was recorded onto the video camera system's hard-drive and an external hard-drive for analysis. Both the vessel and camera video system (SeaTrack^{TM}) recorded the location and track positions along each transect line by dGPS in real-time every second. Mean water depth for each transect line was also recorded from the vessel's on-board acoustic echo depth sounder. I repeated this method along each transect line within the two square survey areas for each temporal sampling period.

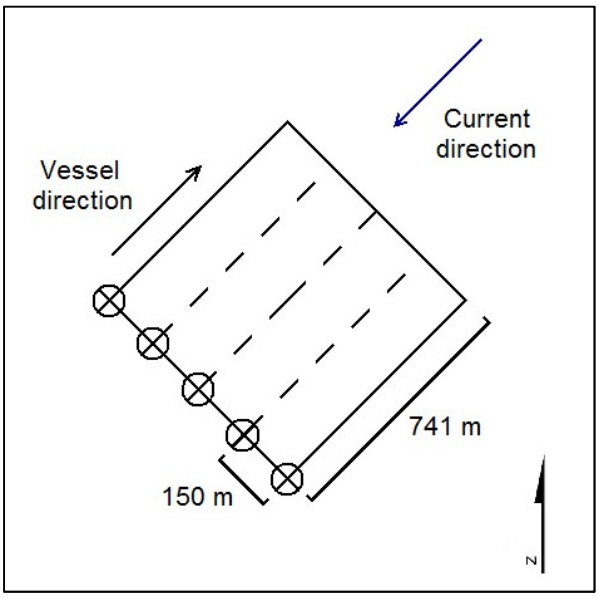


Figure 2.2. Schematic representation of the benthic towed video camera system method within the T60 and T61 survey squares, Alderney Race. Circles represent the starting points of each survey transect line (n = 5), with the camera system deployed from a vessel controlled to drift against the water current direction (0.4 – 2 knots). The distances between each transect and lengths are given in metres (m).

Surveys were conducted during neap tidal cycles for each temporal sampling period. This was to reduce equipment movement, breakage or loss that is more likely to occur during spring tidal cycles in this type of environment. All surveys were undertaken two hours after local high tide as this is known as the slack tidal period within this area of the Alderney Race.

Photographic still analysis

Photographic still images were then extracted from the video camera system's video data using Adobe Photoshop software (Adobe 2012). A total number of twenty images were selected from each transect line within each survey square during every temporal sampling period, to give a total of 1200 images. Images were chosen randomly, based on the image's dGPS decimal minute/ second coordinates, which were selected by number generation techniques in R (based on the minute coordinate number selection ranging 0 - 60) (Service and Golding 2001; MESH 2008; R Development Core Team 2010).

I visually assessed each photograph in terms of their physical substrate and ecological epifaunal proportions. The physical substrates were classified following a generalised version of the Wentworth 1922 sediment scale, which also included an organic material category and the overall ecological epifauna (Table 2.1; Greene et al. 1999; Howell 2010). The ecological epifauna (referred to below as benthic cover species) were identified to species where possible using authorative keys (Hayward and Ryland 1995; Ackers et al. 2007). Apart from *Tubularia indivisa*, hydroid species were difficult to identify during the analysis and were therefore grouped together. Identification of Porifera species are also tentative, as further microscopic identification may sometimes be required (Hayward and Ryland 1995). The proportion of each physical and ecological epifauna category was measured in terms of their total percentage cover (%) within each photographic still frame. This was measured using the 'ImageJ' photographic software, with the photograph still image dimension area approximately 1.25 m^2 (ImageJ 2012).

I then visually assessed each photographic still for individual counts of other motile or individual species (referred to below as singleton species) such as crustaceans, which were also identified to species where possible. I further classified both the benthic cover species and singleton species into feeding regime types for analysis, using key reference materials and online data resources, such as the BIOTIC online database (Table 2.1, MarLIN 2006).

Table 2.1. Physical substrate, ecological and feeding regime type categories used during the 2009 – 2010 Alderney Race benthic physical and ecological study.

Physical substrate categories	Ecological categories	Feeding regime categories
Bare bedrock	Benthic cover species	Omnivore
Boulder (grain size 0.25 - 3 m)	Singleton species	Predator
Cobbles (grain size 64 – 256 mm)		Suspension feeder
Pebbles (grain size 4 – 64 mm)		
Gravel (grain size 2 – 4 mm)		
Sand (grain size < 2 mm)		
Organic material		
Ecological epifauna		

Data analysis

All univariate analyses were undertaken using the R statistical software (R Development Core Team 2010), with all multivariate analyses carried out in the Primer v6 software (Clarke and Gorley 2006).

The physical substrate and ecological benthic cover species percentage cover information from each photograph were averaged across each transect line for each survey square (T60 and T61) and temporal sampling period (February 2009, May 2009, August 2009, November 2009, February 2010 and May 2010) separately for analysis (total number of samples = 60). The total number of singleton species individuals (n) were then calculated for each transect line, within both survey square, across the temporal sampling periods separately (total number of samples = 60).

To identify the benthic physical substrate composition within both survey squares, I determined the mean proportion of each physical substrate category and their standard deviation (across all transect samples for both squares). To examine if there were substrate composition differences between the two survey squares and temporal sampling periods, I used separate one-way analysis of similarities. This was carried out using the Primer v6 ANOSIM routine, with the survey square and temporal sampling period selected as separate factors (Langhamer et al. 2009). This routine is based on a generated Euclidean distance resemblance matrix of the substrate compositional dataset, which was normalised prior to analysis. This resemblance coefficient was selected due to its usefulness for mixed environmental datasets, and the transformation based on visual interpretation of generated

draftsman plots in Primer v6 (Clarke and Warwick 2001; Clarke and Gorley 2006). The ANOSIM routine produces a global R value and corresponding pairwise comparison values, to compare significance between selected groups across samples (Clarke and Gorley 2006). To visually examine the overall relationship of these substrates between the survey squares and temporal sampling periods, I used a hierarchical cluster analysis dendogram. This was carried out using the Primer v6 CLUSTER routine, using group average linkage.

I used generalised linear models (GLMs) to look for differences between the survey squares and temporal sampling periods in: i) percentage cover of benthic cover species (%), ii) the abundance of singleton species (n) and iii) their diversity separately, using species richness (*S*), Shannon-Wiener diversity (H', $\log_e$) and Pielou's evenness (J') diversity measures. These diversity measures were generated from the Primer v6 DIVERSE routine. The benthic cover species percentage cover GLM used a binomial distribution, with the singleton abundance (n) and diversity (*S*) GLMs using a quasi-poisson distribution. The quasi-poisson distribution error structure was used as these datasets showed over-dispersion (Crawley 2007). The Shannon-Wiener and Pielou's evenness diversity measures were assessed using a Gaussian distribution. Minimum adequate models (MAMs) were produced, following term deletion methods from the maximal models, which included all variable interaction terms (Crawley 2007). This involved removing non-significant variables sequentially (and their interaction) using ANOVA (with appropriate F or Chi tests), to justify excluding variables (where $p > 0.05$), to compare model fit (Crawley 2007). All models were visually assessed for heterodascity and non-normality by Q-Q normality plots, with no transformations selected.

The composition of benthic cover species, singleton species and their separate feeding regimes were then compared between the two survey squares and temporal sampling periods. This was assessed using separate one-way ANOSIM routines, with the survey square and temporal sampling period selected as exploratory factors. I then applied one-way SIMPER routines to each dataset, to identify the percentage contribution of each species and feeding regime type, within each survey square and temporal sampling period. This routine decomposes all average Bray-Curtis similarities among samples within a group into percentage contributions (%) from each variable (Clarke and Warwick 2001). This was based on a 100 % cut-off point of the total percentage contribution within each sample, due to the overall small number of observed variables recorded during the study (Clarke and Gorley 2006).

For benthic cover and singleton species, I then further examined their overall compositional similarity between the survey squares and temporal sampling periods, using a cluster analysis dendogram (with group average linking in the Primer v6 CLUSTER routine). Prior to all analyses, the benthic cover species and corresponding feeding regime datasets were square root transformed, with the Bray-Curtis similarity resemblance matrix selected for analysis. This transformation was selected to moderately weight the contributions of common and rare species for these multivariate analyses. The singleton species and their corresponding feeding regimes were fourth root transformed, with the Bray-Curtis similarity resemblance matrix selected, which comprised of with an added (+1) dummy variable. This dummy variable and transformation was selected due to the low abundance within the overall singleton species dataset (Clarke and Gorley 2006).

To examine the relationship between the physical substrate and ecological compositions, I used the Primer v6 biota and environment matching analysis, BEST procedure (BIO-ENV option). This routine compares the multivariate physical and ecological resemblance matrices, using Spearman rank correlation coefficients (ρ) (Clarke and Gorley 2006). The highest ranked correlation for each combination of physical variables are displayed (K), which are responsible for the biotic patterns observed. The global BEST permutation match test analysis was included in this analysis, to examine the overall significance of the relationship between the physical and ecological multivariate resemblance matrices ($\rho = 0$, represents the null hypothesis, ran on 99 permutations). I ran these analyses separately for the benthic cover species (square root transformation, Bray-Curtis resemblance) and singleton species (fourth root transformation, Bray-Curtis resemblance) similarity resemblance matrices with the physical substrate (normalised, Euclidean distance resemblance) similarity resemblance matrix for both survey squares.

2.4. Results

The study recorded a total number of 7 physical substrate categories and 19 benthic species. Benthic species comprised of 11 cover species, which were represented by 1 Bryozoa, 1 Crustacea, 2 Hydrozoa, and 7 Porifera species. A total number of 8 singleton species were recorded (65 individuals overall), which comprised of 5 Cnidaria, 2 Crustacea and 1 Echinodermata species respectively (see Appendix 1 Table A.1 for species lists). These

species were then classified into three feeding regime types; suspension feeder, omnivore or predator types.

Physical substrate composition

The photographic stills showed a range of physical substrate characteristics within the T60 and T61 survey squares of the Alderney Race (Table 2.2). The ANOSIM results outlined that the composition of these substrates were highly comparable between the two squares (R = 0.024, p = 0.31), comprising of gravels, sands and cobbles. The cluster analysis showed the overall variability of substrate composition between the survey squares and temporal sampling periods (Figure 2.3). The analysis revealed the most dissimilarity of substrate composition during February 2010 within the T61 survey square and February 2009 within the T60 square. This was further shown by the weak compositional differences between the temporal sampling periods outlined by the ANOSIM results (R = 0.3, p = 0.045).

Table 2.2. Mean benthic physical substrate type and composition identified within the T60 and T61 survey squares, from February 2009 – May 2010, Alderney Race. Substrate proportions are given as mean percentage cover (%) and standard deviations (SD) of both physical characteristics and ecology (species percentage cover), across all temporal sampling periods, for both squares derived from the towed video camera photographic stills.

Substrate categories	**Location:** **T60**		**T61**	
	Mean (%)	**SD**	**Mean (%)**	**SD**
Physical characteristics				
Bare bedrock	6.4	4.58	10.1	8.83
Boulder (grain size 0.25 - 3 m)	1.41	1.62	2.91	4.52
Cobbles (grain size 64 – 256 mm)	16.16	10.48	6.58	5.78
Pebbles (grain size 4 – 64 mm)	5.41	11.42	2.41	3.74
Gravel (grain size 2 – 4 mm)	18.79	18.1	17.40	17.1
Sand (grain size < 2 mm)	11.58	18.32	12.83	15.75
Organic material	0	0	0.04	0.10
Ecological epifauna	39.88	13.22	47.68	18.68

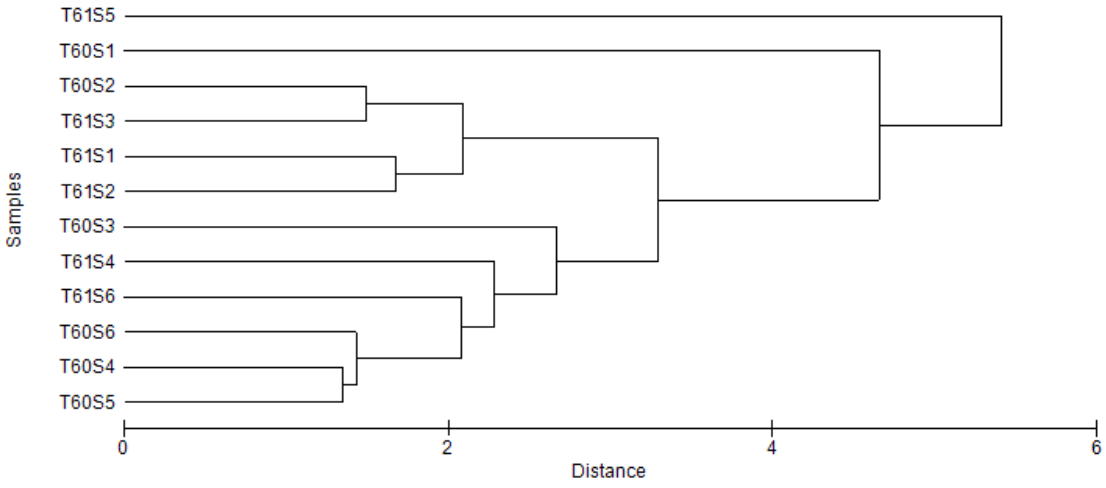


Figure 2.3. Hierarchical cluster analysis dendogram of the benthic physical substrate composition within the T60 and T61 survey squares, for each temporal sampling period (S1 – S6), from February 2009 – May 2010, Alderney Race. Lettering corresponds to each temporal sampling period: S1 = February 2009, S2 = May 2009, S3 = August 2009, S4 = November 2009, S5 = February 2010, S6 = May 2010.

Benthic species abundance and diversity

The proportion of benthic species percentage cover (Table 2.2) and singleton species abundances (mean n: T60 = 5.50; T61 = 5.33) were both similar between the T60 and T61 survey squares (Table 2.3).

I also found comparable species richness (*S*) estimates between the two survey squares (Table 2.3) for benthic cover species (mean *S*: T60 = 6.16; T61 = 5.83) and singleton species (mean *S*: T60 = 2; T61 = 1.66). This was also identified with the Shannon-Wiener diversity measure for both benthic cover species (mean H': T60 = 1.62; T61 = 1.58) and singleton species (mean H': T60 = 0.53; T61 = 0.37). In addition, Pielou's evenness estimates were also similar for benthic cover (mean J': T60 = 0.916; T61 = 0.924) and singleton species (mean J': T60 = 0.474; T61 = 0.370) between the two survey squares.

Across the six temporal sampling periods, benthic species mean percentage cover and singleton species abundance showed comparable estimates within the two survey squares (Table 2.3; Figure 2.4; Figure 2.5). Both benthic cover species and singleton species' richness

and Shannon-Wiener diversity measures however, significantly fluctuated throughout the study, within both survey squares. Overall, benthic cover species comprised of similar high estimates of Pielou's evenness across the study. Singleton species evenness fluctuated across the study, yet was similar across the two squares overall, due to the low numbers of species observations actually recorded.

Table 2.3. Generalised linear minimal adequate model results (MAM) for benthic cover species a) and singleton species b) proportion cover (%), abundance (n) and diversity measures (*S*, H' (log_e), and J') within the T60 and T61 survey squares, from February 2009 – May 2010, Alderney Race. F represents the overall F model value; P represents the model P value; DF represents the degrees of freedom; n.s. represents no significant explanatory variables (n = 60).

	Model explanatory variable	F value	P	DF
a) Cover species	**Species cover (%)**	n.s		
	Species richness (*S*)			
	Intercept			
	Temporal sampling period	12.70	0.003	54
	Shannon-Wiener diversity (H')			
	Intercept			
	Temporal sampling period	17.39	0.001	54
	Pielou's evenness (J')	n.s		
b) Singleton species	**Species abundance (n)**	n.s		
	Species richness (*S*)			
	Intercept			
	Temporal sampling period	2.71	0.01	54
	Shannon-Wiener diversity (H')			
	Intercept			
	Temporal sampling period	7.46	0.01	54
	Pielou's evenness (J')	n.s		

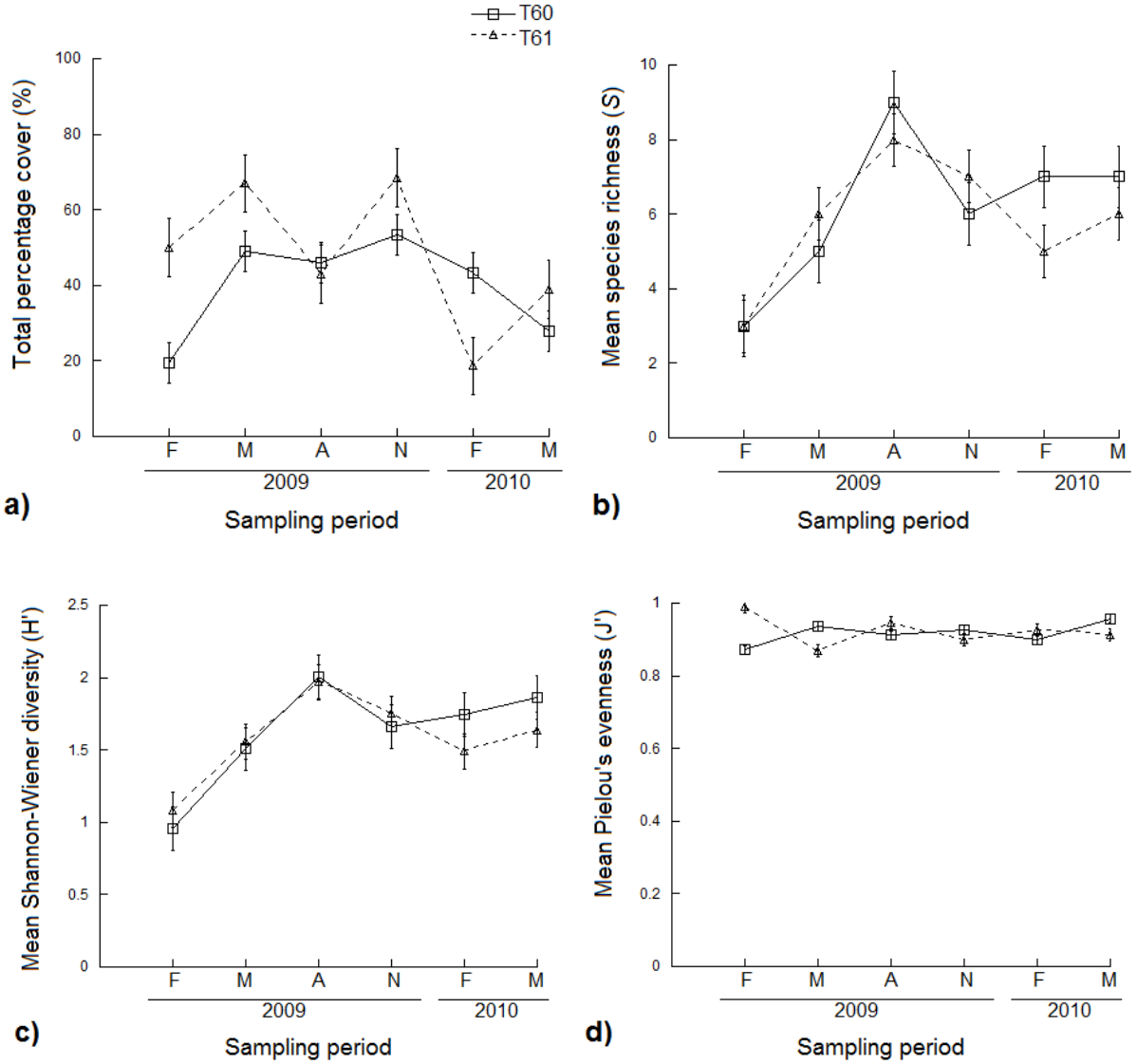


Figure 2.4. Benthic cover species total percentage cover a), mean species richness (*S*) b), Shannon-Wiener (H', $\log_e$) c), and Pielou's evenness (J') diversity measures d), within the T60 and T61 survey squares, from February 2009 - May 2010, Alderney Race. Solid line represents the T60 square, the dotted line represents the T61 square (± 1 SEM). Letters represent the temporal sampling periods: F = February, M = May, A = August and N = November.

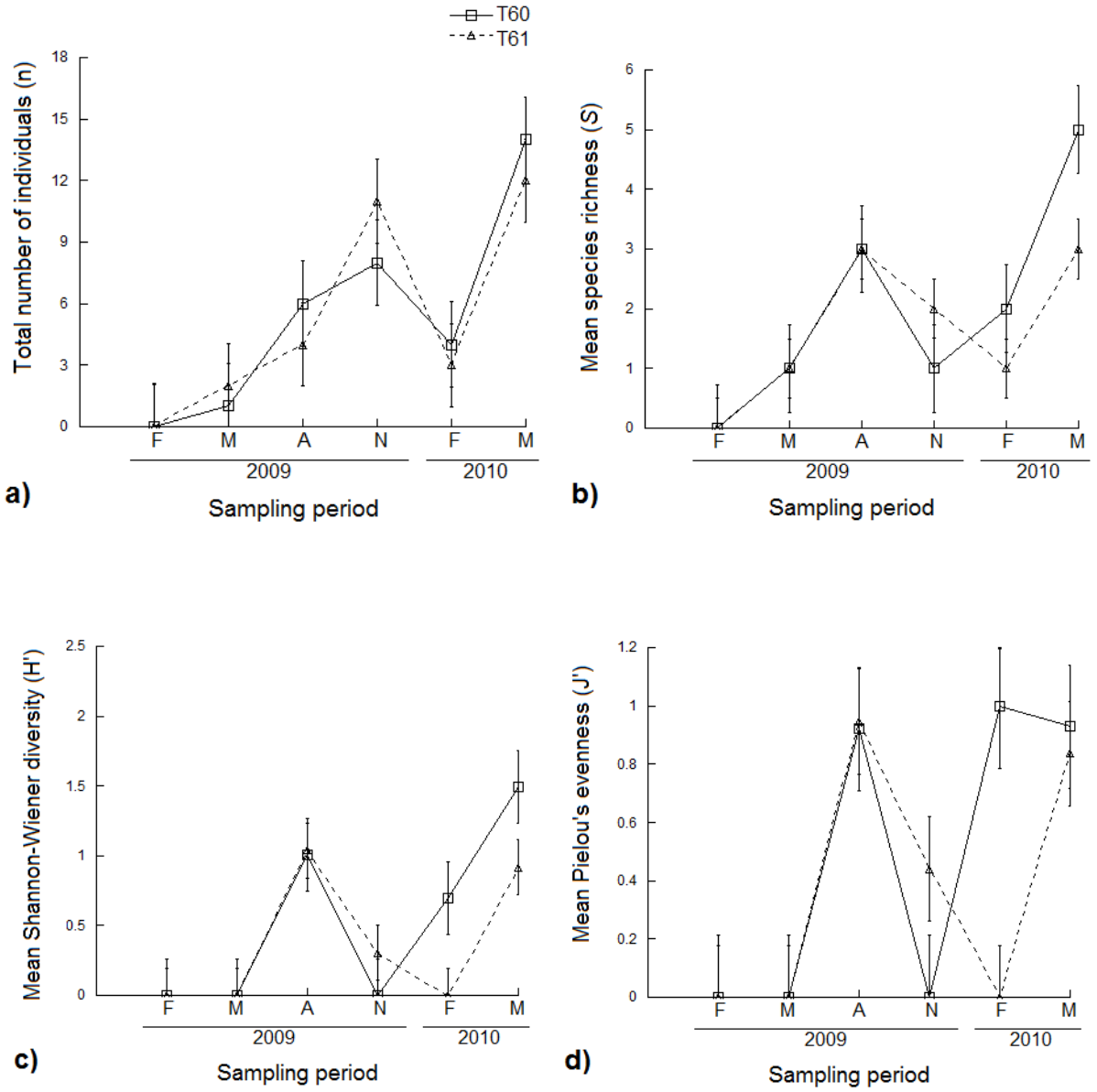


Figure 2.5. Benthic singleton species total abundance (n) a), mean species richness (*S*) b), Shannon-Wiener (H', $\log_e$) c), and Pielou's evenness (J') diversity measures d), within the T60 and T61 survey squares, from February 2009 - May 2010, Alderney Race. Solid line represents the T60 square, the dotted line represents the T61 square (± 1 SEM). Letters represent the temporal sampling periods: F = February, M = May, A = August and N = November.

Benthic species composition

The ANOSIM results identified that the composition of benthic cover species were highly comparable between the two survey squares (R = -0.11, p = 0.87). From the SIMPER results (Table 2.4), I found that the survey squares predominantly comprised of the crustacean *Balanus crenatus*, and the hydroid, *Tubularia indivisa*, with all cover species consisting solely of the suspension feeding type (ANOSIM: R = -0.08, p = 0.73). The ANOSIM and SIMPER results also showed similar singleton species compositions between the two squares (R = -0.146, p = 0.95), which was largely represented by the two cnidarians; *Urticina felina* and *Alcyonium digitatum*, and the crustacean, *Cancer pagurus*. Overall, the feeding regimes of the recorded singleton species were similar between the two survey squares (ANOSIM: R = -0.142, p = 0.92), and represented by large proportions of omnivores and predators.

The ANOSIM and SIMPER results revealed significant differences of the composition of the benthic cover species across the temporal sampling periods (R = 0.55, p = 0.004; Table 2.5). The cover species cluster analysis further revealed these three distinct communities, between: February 2009 and 2010; May 2009; and the other 2009 and 2010 May, August and November temporal sampling periods (Figure 2.6a).The SIMPER results showed that during the February 2009 and 2010, the survey squares primarily comprised of *B. crenatus* and *T. indivisa* species, whilst the May 2009 comprised of additional Porifera species. The other 2009 and 2010 temporal sampling periods consisted of a larger range of bryozoan, Porifera and hydroid species overall. Across the temporal sampling periods, the benthic cover species suspension feeding regime was highly similar (ANOSIM: R = 0.039, p = 0.33).

The ANOSIM analyses also outlined that singleton species composition altered across the temporal sampling periods (R = 0.689, p = 0.001). The singleton species cluster analysis showed these two distinct communities across the temporal sampling periods (Figure 2.6b). This was between: February and May 2009 and 2010, and, the other 2009 and 2010 May, August and November sampling periods. The SIMPER results here showed that the February and May 2009 and 2010 samples were represented by no species observations or primarily *A. digitatum* (Table 2.5). I also found that the other 2009 and 2010 temporal sampling periods were represented by *U. felina*, *C. pagurus* and *Homarus gammarus* species. Singleton species feeding regime type altered across the temporal sampling periods (ANOSIM: R = 0.722, p = 0.002), with omnivores found throughout the temporal sampling periods and predators specifically within May 2009.

Table 2.4. Benthic cover a) and singleton species b) presence and one-way SIMPER percentage contribution species composition and feeding regime (%) results, within the T60 and T61 survey squares, from February 2009 – May 2010, Alderney Race. The presence of each species within each survey square is represented by +. Bold indicating ≥ 20 %, with average similarity representing the SIMPER contributions across all samples.

	Taxonomic categories	Presence		SIMPER (%)	
		T60	T61	T60	T61
a) Cover species					
Bryozoa	*Flustra foliacea*	+	+	17.15	9.33
Crustacea	*Balanus crenatus*	+	+	**28.80**	**31.30**
Hydrozoa	*Hydrozoa spp*	+	+	3.28	10.50
	Tubularia indivisa	+	+	**24.87**	**32.05**
Porifera	*Amphilectus fucorum*	+	+	2.09	0.52
	Cliona celata	+	+		1.16
	Halichondria (Halichondria) panicea	+	+	17.58	10.85
	Hymeniacidon perlevis		+		
	Myxilla (Myxilla) incrustans	+	+	1.34	
	Pachymatisma johnstonia	+	+	4.89	4.29
	Tethya citrina	+	+		
Average similarity				59.55	54.71
Feeding regime	Suspension Feeder	+	+	**100**	**100**
Average similarity				88.82	86.46
b) Singleton species					
Cnidaria	*Actinothoe sphyrodeta*		+		
	Alcyonium digitatum	+	+	12.02	**26.52**
	Corynactis viridis	+			
	Metridium senile		+		
	Urticina felina	+	+	**65.31**	**56.22**
Crustacea	*Cancer pagurus*	+	+	**22.68**	17.26
	Homarus gammarus	+	+		
Echinodermata	*Asterias rubens*	+			
Average similarity				24.63	12.67
Feeding regime	Omnivore	+	+	**74.87**	**89.07**
	Predator	+	+	**25.13**	10.93
Average similarity				37.43	30.76

Table 2.5. One-way SIMPER percentage contribution (%) results of benthic cover a) and singleton species b) composition and feeding regime composition across each temporal sampling period, from February 2009 – May 2010, Alderney Race. Bold indicating ≥ 20 % and average similarity representing the SIMPER percentage contributions across all samples. – represents zero observations, * representing all SIMPER similarities are zero.

	Taxonomic categories	**Sampling period**					
		Feb 2009	**May 2009**	**August 2009**	**Nov 2009**	**Feb 2010**	**May 2010**
a) Cover species							
Bryozoa	*Flustra foliacea*		**37.76**	8.67	**21.18**		15.33
Crustacea	*Balanus crenatus*	**52.42**		**31.67**	**39.21**	**43.88**	**24.23**
Hydrozoa	*Hydrozoa spp*			10.81		19.62	10.84
	Tubularia indivisa	**31.01**	**42.22**	12.92	**22.64**	**21.94**	**28.67**
Porifera	*Amphilectus fucorum*		6.67	4.07			
	Cliona celata			8.67			
	Halichondria (Halichondria) panicea	16.58		15.02	16.98	14.55	7.66
	Pachymatisma johnstonia		13.35	8.17			13.27
Average similarity		64.01	68.84	83.63	72.06	61.94	69.86
Feeding regime	Suspension Feeder	**100**	**100**	**100**	**100**	**100**	**100**
Average similarity		76.89	92.19	98.37	93.87	79.31	91.73
b) Singleton species							
Cnidaria	*Alcyonium digitatum*		**100**				
	Urticina felina			**45.68**	**100**		**42.78**
Crustacea	*Cancer pagurus*			**54.32**			**28.61**
	Homarus gammarus						**28.61**
Average similarity		-	91.36	65.40	75.42	*	67.37
Feeding regime	Omnivore			**100**	**100**	**100**	**100**
	Predator		**100**				
Average similarity		-	91.36	54.71	75.42	96.41	72.87

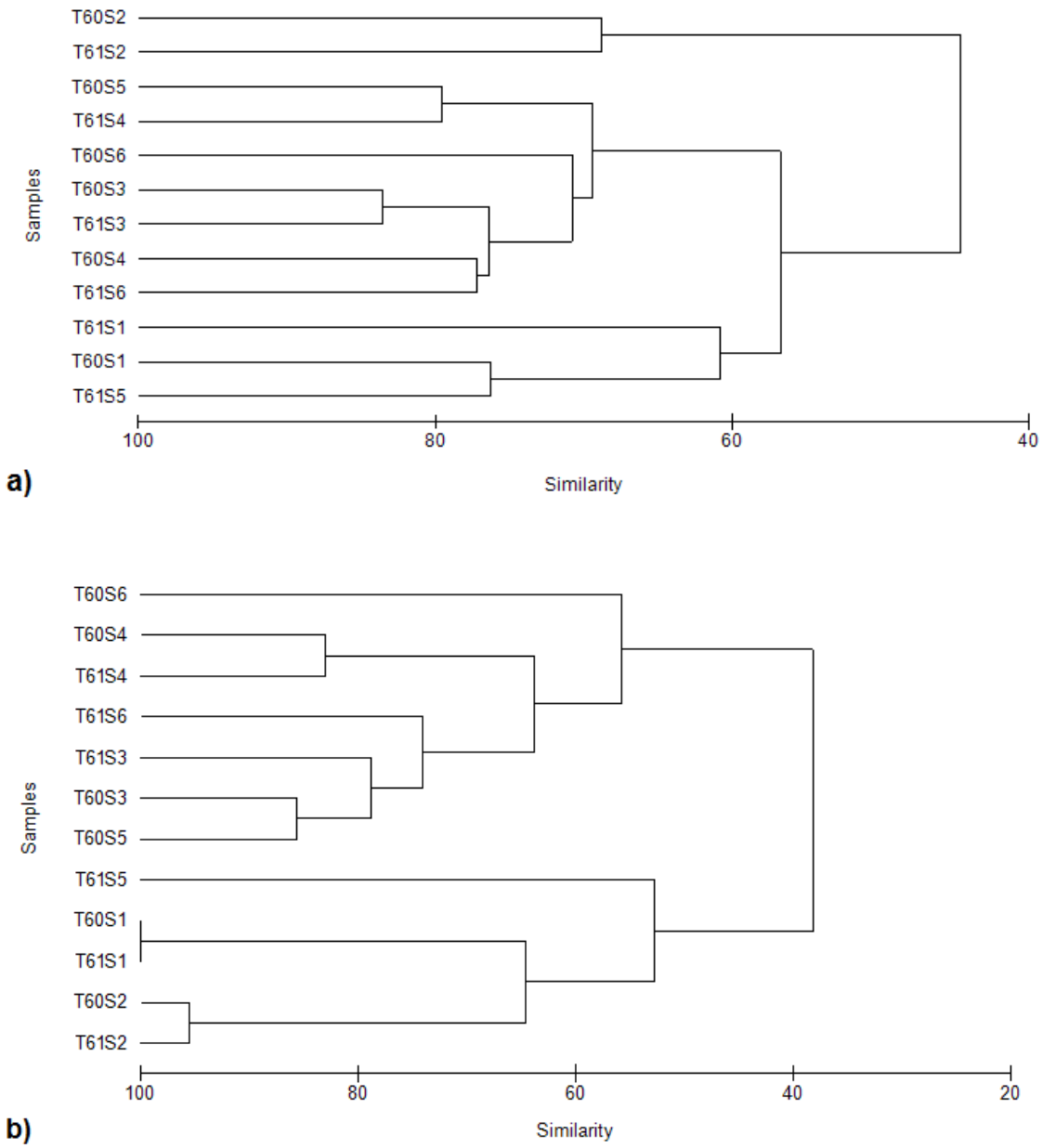


Figure 2.6. Hierarchical cluster analysis dendograms of the benthic cover species a) and singleton species b) compositions within the T60 and T61 survey squares, for each temporal sampling period (S1 – S6) from February 2009 – May 2010, Alderney Race. Lettering corresponds to each square temporal sampling period: S1 = February 2009, S2 = May 2009, S3 = August 2009, S4 = November 2009, S5 = February 2010, S6 = May 2010.

Physical and biotic composition associations

The global BEST match test routine showed no significant relationships between the benthic cover species (T^{60}: rho = 0.739, p = 0.12; T^{61}: rho = 0.794, p = 0.09) and singleton species (T60: rho = 0.679, p = 0.06; T61: rho = 0.446, p = 0.4) with the physical substrate types, for both survey squares. The BIO-ENV exploratory routine further showed a range of substrates combinations affiliated with the cover and singleton species such as pebbles, rocks and boulders (Table 2.6).

Table 2.6. BIO-ENV results for benthic cover a) and singleton species b) correlated with abiotic substrate factors identified within the T60 and T61 survey squares, from February 2009 – May 2010, Alderney Race. *K* represents the total number of factors used to derive the highest Spearman rank correlation coefficients (ρ). Letters represent abiotic substrate types: a = sand, b = gravel, c = pebbles, d = cobbles, e = rocks, f = bedrock, g = organic material. Bold indicates the highest rank correlation in each survey square, for cover and singleton species.

	Location:			
	T60		**T61**	
K	**Correlation (ρ)**	**Factors**	**Correlation (ρ)**	**Factors**
a) Cover species				
1	0.649	c	**0.794**	**c**
2	0.720	c, f	0.493	b, c
3	**0.739**	**c, e, f**	0.414	b, c, f
4	0.704	c-f	0.189	b-d, f
5	0.643	a, c-f	0.06	a-c, e, g
6	0.576	All	-0.043	a-d, f, g
7	-		-0.129	All
b) Singleton species				
1	0.55	c	0.327	g
2	0.633	c, f	0.431	c, g
3	0.668	a, b, f	0.435	c, e, g
4	**0.679**	**a, b, d, f**	**0.446**	**c, e-g**
5	0.668	a, b, d-f	0.332	c-g
6	0.589	All	0.225	a, c-g
7	-		0.121	All

2.5. Discussion

Benthic physical characteristics

The physical properties identified during this study are generally comparable to previous geological bathymetry mapping information and research of the Alderney Race environment, at these measured scales (Osiris Projects 2009; Coggan and Diesing 2011). The survey area is characterised by complex topography and geomorphology, which is represented by the wide range of physical substrates I recorded throughout this study. In addition, the study identified seafloor characteristics such as gravels, sands and organic material, which were not recorded in previous surveys. These previous surveys used sampling methods such as Side Scan Sonar (SSS) or Multibeam Echosounder Systems (MBES), to identify seafloor characteristics at broader spatial scales (> 1km). Therefore, adopting a small scaled approach prior to, or part of, wider-scaled surveys, may produce more accurate bathymetry information for this complex environment. For example, highly detailed bathymetry seafloor maps could be created using both a small and broad scaled approaches, which would be useful for ecological and future renewable energy development purposes or marine spatial planning activities.

My results show some temporal variation in the composition of these substrates, which might be related to the environment's known strong hydrodynamic regimes (Neill et al. 2012). Hydrodynamic regimes (tidal currents and wind driven waves) are the main drivers of temporal modifications in geological bed-form, and sediment transport processes, such as scour (Callaway et al. 2009; Coggan and Diesing 2011). Temporal variations can include annual, seasonal and monthly scales, with the strongest hydrodynamic regimes found during episodic winter storms and 'spring' tidal cycles (van Rein et al. 2011). Storlazzi et al (2011) describe that complex systems, such as the Alderney Race, can temporarily comprise of fine grain sediments during periods of low hydrodynamic regimes. Substrates are then subsequently removed as the energetic properties of the present hydrodynamic regimes begin to increase.

Although this study comprised of few temporal sampling periods (n = 6), my observations are compatible with this pattern, with the presence of fine grain sediments observed in the months (sampling periods: May and February in cluster analysis and see Appendix 2 Tables A.2.3-2.4 for further substrate proportions) following possible winter storm events or 'spring' seasonal tide cycles (Neill et al. 2012). Further temporal sampling will be required to further determine these environmental processes (Brown et al. 2011). In addition, each sampling

period was always conducted during the 'neap' period of the tidal cycle. An additional study of the temporal pattern of these physical characteristics within a full tidal cycle (spring and neap scenarios) would give a more in-depth review and in turn, their influence on the surrounding ecology at a finer-scale.

Benthic ecology

The highly spatial similarity of the ecology identified during this study, allow for these survey squares to be considered as reference sites for benthic assemblages within tidal race environments. Such sites are of particular importance where limited records exist, for future natural and anthropogenic comparisons (Rodrigues et al. 2011). These two sites are characterised by few species (particularly singleton species), which show morphological physical resilience, and life history strategies, associated to environments with intense hydrodynamic regimes (Hiscock 1983; Palumbi 1984; Wildish et al. 1987; Fegley et al. 1992; Connor et al. 2004). For example, the study recorded species such as *Tubularia indivisa* and *Halichondria* (*Halichondria*) *panicea*, which are indicative of increased tidal flow conditions (Hiscock 1983). This follows past research, which describe hydrodynamic forces, such as maximum velocity rate and seabed shear stress, to heavily influence marine communities (both directly and indirectly) in terms of presence, distribution and behaviour (Warwick and Uncles 1980; Sobral and Widdows 2000; Rodrigues et al. 2011). For example, benthic cover Hydrozoa and Porifera species are often related to high tidal velocities rates, which can increase the availability of suspended particles and re-suspended materials for feeding (Flach et al. 1998; Okamura and Partridge 1999).

These results also show significant temporal changes (species and group assemblages), with overall species diversity initially increasing after the winter periods, across these short-term temporal sampling periods (February sampling periods in cluster analysis). Similar temporal patterns have been recorded across a variety of marine environments, including estuarine, intertidal, shallow sublittoral and deep-sea regions (Foster 2003; Catalán et al. 2006; Post 2008; Wernberg and Goldberg 2008). Causal factors are related to the temporal shifts and interactions with a range of environmental and biotic processes, including sea surface temperature, irradiance, recruitment and reproduction (Ordines et al. 2011; Post et al. 2011). Okamura and Partridge (1999) also identified that during winter and early spring periods, marine species are particularly susceptible to increased intensity of hydrodynamic regimes

and frequency of storm events. Such processes can (directly) damage the physical form (abrasion or complete removal) of marine organisms, or (indirectly) reduce their ability to feed effectively and also increase post-larvae settlement mortality (Hiscock 1983). This is shown in my results, through the altered assemblage compositions between the February and May seasons. Periods of reduced hydrodynamic regimes, such as the summer periods (such as the August season), are therefore more favourable for biotic interactions, including growth and predation (Wildish et al. 1987; van Rein et al. 2011). Overall, this suggests that benthic assemblages within this tidal race environment, not only undergo intense hydrodynamic regimes on a daily basis, but are also influenced from a complex range of abiotic and biotic factors, which change temporally. As such, it further highlights the need for further temporal investigations (spring/neap cycles; seasonal and annual scaled assessments) and direct and indirect effects upon the ecology within this environment type.

The benthic cover hydrozoan species, *T. indivisa*, which I frequently observed during the survey, is acknowledged as a pioneer, short-lived, and often associated with disturbed benthic environments (Zintzen et al. 2008). Interestingly, Zintzen et al (2008) also report that during temporal investigations, proportions of this species declined after summer periods. This was related to factors such as increased predation, or a lack of available recruits. This pattern has been observed in a number of other studies, and suspension feeding type species (Coma et al. 2001; Balata et al. 2006). My findings are different to these studies, and show increased *T. indivsa* recorded after the August, 'summer' sampling period. This may be due to the causal factors described above, or alternatively, related to the increased hydrodynamic regimes that can occur within this environment during the autumn periods (sampling periods: August and November). As this species can be regarded as an early coloniser, its increased presence may be more related to the temporal process of disturbance from these increased hydrodynamics periods (Tilman 1990). Overall, this outlines that even during the temporal shift of such extreme environmental processes, tidal race environments, can still sustain certain species over time. This has fascinating implications to the ecological interactions and assemblage characteristics that can occur within them, such as the role of predation and succession (Dayton 1971; Sousa 1979).

Further methodological environmental site and temporal comparisons between these studies would be extremely useful to examine these patterns, biotic interactions and causal factors in greater detail. Nevertheless, this fine-scale approach begins to highlight the processes that

occur within these unknown environments, in terms of species recruitment, settlement and disturbance, over time.

Combined physical characteristics and ecology

Due to the topographic complexity of the seabed, variety of recorded substrates and the potential for scour, it seems probable that the physical characteristics measured here are poorly related to the observed benthic assemblages. Studies conducted in other, similar dynamic environments suggest that these assemblages are more strongly related to factors such as tidal bed stress, velocity rate, Chlorophyll *a* concentrations and bottom temperature, which should be considered for future surveys (Bolam et al. 2010; Gogina et al. 2010; Rodrigues et al. 2011; Burrows 2012). The inclusion of other environmental variables in this study may enable a more definitive view of the role (direct and in-direct effects) and relationship between the present hydrodynamic regimes and the ecology within these environments. These studies however, used generalised modelling methods to measure such variables i.e. broad-scaled tidal velocity models or generalised descriptions. As mentioned earlier, these techniques may be inappropriate for these complex bathymetry systems, with finer scale variable monitoring and field measurements suggested; for greater accuracy prior to such broad-scale assessments (Bailly du Bois et al. 2012). Therefore, future studies should consider the integration and selection of new variables carefully, particularly with respect to the tidal race environment itself.

The overall physical characteristics and ecology recorded during this study could, however, aid European marine habitat classification schemes, such as EUNIS (European Nature Information System). Such classifications have limited habitat records within these types of environments, and the Channel Islands as a whole (Connor et al. 2004). Using these classification schemes, the study sites here could be regarded as the '*A.4.11: very tide swept faunal communities on circalittoral rock'* habitat complex, based on their comparative physical and species compositions (EEA 2004). My results therefore provide supplementary information for this habitat classification, in terms of the compositional and temporal patterns of the physical characteristics and ecology (Bremner et al. 2006). Though, further verification of my results with this habitat type is essential, as circalittoral environments are difficult to satisfactory classify (Connor et al. 2004). Furthermore, this verification should also involve regional comparisons to other tidal race environments, such as the Big Russell, Guernsey or

the Pentland Firth, Scotland. This may aid the creation of a sub-habitat tidal race environment category, due to the present natural rarity of this environment, the lack of records and also the generality of the habitat descriptions. It would enable more suitable habitat classification of these environments on a wider- geographically scale which would be extremely useful for natural and anthropogenic impact assessments.

Future considerations and conclusions

This study presents a fine-scale assessment of the benthic physical characteristics and ecology of the Alderney Race tidal race environment. The environment represents a complex system, influenced by a potential range of factors, which change through time. To build upon these initial results, future studies could expand to other, comparable locations (in terms of water depth) within the total extent of the Alderney Race. This would increase the validity of these results in terms of replication, and baseline knowledge of these depths at a wider, spatial scale. Future studies could then consider extending to different water depths locations (water depths 10-20 metres), and topographic locations (pinnacles and sandbanks) to understand this complex environment as a whole. These new studies could also consider extending the temporal scales that were used during the study. As described earlier, the measured seasonal scales were only conducted during neap tide cycle periods. Therefore measuring full tidal cycles (both spring and neap tidal cycles), and other temporal scales (longer termed seasonal and annual surveys) may further define physical and ecological (and their relationship) information within this uncommon environment. Furthermore, any future study considerations would also benefit from the addition of supplementary physical variables. The addition of other direct physical measurements, such as tidal velocity rate, bed sheer stress and longer-term tidal amplitude information would help further define relationships between the physical and biotic patterns, and aid benthic habitat descriptions (Rodrigues et al. 2011).

In addition to this, I would also propose further video camera method testing within this environment. The camera system used here was particularly valuable during the field campaign, in terms of durability and ease of use within this extreme environment. The derived results were also useful for outlining the primary characteristics within the study sites, and corresponded well to previous research conducted within the area (Osiris Projects 2009). During the post-processing of the video and photographic analysis stage, it was

however, difficult to identify species, with certain species identified to a coarser taxonomic level i.e. benthic cover bryozoan and hydrozoan turf species or missing other singleton species. As such, the results shown here may not represent or bias the overall diversity estimates and compositional patterns of the ecological properties observed within this system. Although this is a common problem with this type of surveying methods, on-going method testing during the proposed future studies would inherently increase sampling accuracy and data validation, especially for under-sampled environments such as this. This may also include focussing upon the total proportion of cover species (i.e. potentially removing species identification bias), selecting specific species for study (i.e. *Balanus crenatus*) or re-evaluating the use of cover and singleton species for benthic ecological assessments used here. I would also recommend greater collaboration and method comparisons with other available information, schemes or research groups that use such methods within these types of environments such as the MNCR, PRIMARE, IFREMER and the JNCC EuSeaMap scheme to enhance current surveying techniques (Burrows 2012).

Yet, these future studies may still face the difficulties of sampling within this type of extreme offshore environment. The sampling effort during this study was reduced, in terms of the spatial extent and measured temporal scale. This was due to the intense hydrodynamic regimes, unpredictable weather conditions and complexity of seabed facies that commonly occur within this system which will therefore be an on-going challenge for any future study design (i.e. wider scaled spatial and temporal scales). Therefore, I primarily recommend the use of a combined approach, with the above future field-based sampling strategies with other complementary techniques, such as computerised environmental variable modelling. A small number of hydrodynamic records and models based on the Alderney Race now exist, from academic sources (Bailly du Bois et al. 2012; Neill et al. 2012) and commercial offshore tidal energy exploitation developers (such as ACRE and ARE Ltd). Combining both field based sampling and computerised applications could potentially reveal environmental and biotic interactions within these environments, aid data validation, and could also lead to a variety of new ecological investigations. For example, these environmental variable models and data sources also comprise of wider-scaled hydrodynamic gradient information (i.e. full tidal velocity scales, < 0.1 - > 4.0 m/s). This information could therefore lead to future predictive benthic cover species distribution modelling studies, whereby relating species presence with tidal velocity rates. Such future studies could enable accurate benthic habitat and species mapping of the Alderney Race, and would be extremely useful for tidal energy development

ecological concerns. Alternatively, such combined approaches could re-examine ecological theories, including the role of disturbance in complex offshore marine environments. This could relate field-based sampling, bathymetry and environmental modelling techniques with the temporal extent, density and vertical profile of colonising species, such as *T. indivisa*. In addition it may further outline direct and indirect influences of the hydrodynamic regimes upon the present assemblages. Ultimately, using a combined approach would allow the extension of the study in terms of scale, method accuracy, measured variables and ecological investigation, whilst reducing the concern for unsuccessful field-based sampling effort. This approach could then be applied to other tidal race environments, to advance knowledge within these under-sampled environments. Overall, such efforts are crucial, to help assess potential natural and anthropogenic impacts that these rare environments may face in the future.

Chapter 3. Intertidal rocky-shoreline ecological variation within a marine tidal race environment.

3.1. Abstract

The intertidal rocky shoreline system is a useful marine environment for ecological investigation and has been the subject of a variety of studies in the past. Despite this, ecological investigation within the intertidal shorelines of marine tidal race environments is rare, partly because they are uncommon. Defining baseline ecological patterns and their variation within this environment type is important due to the potential for future impacts from natural and anthropogenic activities in these intertidal regions. Here I examined the intertidal macroalgae and faunal ecological components within the Alderney Race, Channel Islands. I assessed macroalgae and faunal species diversity, composition and colonisation across two intertidal bays within this environment type. These were measured across different spatial and temporal scales, using field-based sampling methods, from 2009 – 2011.

I found that intertidal macroalgae and faunal species ecological components showed variable spatial and temporal heterogeneity between the two bays within this marine tidal race environment. Species diversity and composition showed different, graduated responses across these shorelines, which I related to underlying extrinsic and intrinsic properties that are commonly found within intertidal systems. The annual process of macroalgae colonisation differed between the two bays, whilst faunal species colonisation was similar across all measured spatial and temporal scales. Overall, these results highlighted the variation of these ecological components between the two bays and their overall use to identify different ecological aspects within environments that are presently under-sampled.

3.2. Introduction

The intertidal rocky shoreline is a key marine environment for ecological study. This system has been used to develop ecological theories or test key hypotheses, due to the varying environmental gradients and biological interactions found within them (Underwood et al. 2000). Consequently, there is now an extensive number of ecological intertidal studies, which range from detecting invasive or indicator species (Juanes et al. 2008; Olabarria et al. 2009; Mineur et al. 2010), anthropogenic impacts and climate change responses (Mieszkowska et al. 2006; Payne et al. 2008), to investigating patterns of species biodiversity and composition (Underwood 2000; Martins et al. 2010). These ecological intertidal studies are commonly designed as either descriptive or manipulative experiments, and use a variety of field-based or modeling methods for research strategy objectives (Underwood et al. 2000).

A common research strategy approach is to compare intertidal rocky shoreline species diversity and compositional patterns across differing environmental gradients, such as hydrodynamic forces, which can include the level of wave exposure, tidal stream velocity intensity or (spatial and temporal) scales. This approach can provide valuable information, to identify ecological responses or interactions with changing environmental processes over time. It includes deriving a range of quantitative ecological and environmental information (including data derived from novel research methods) which can be used to validate and test ecological models and theories (Underwood 2000). For example, studies which use this research strategy approach have frequently shown that the level of exposure (wave intensity or tidal height) and tidal velocity strongly affects species presence, distribution and community structures (Hiscock 1983; Denny 2006; Burrows et al. 2008; Ordines et al. 2011). Hydrodynamic forces can directly influence species within a variety of marine environments, through physical stress and substrate loss, or indirectly via propagule dispersal, food supply, and foraging abilities (Warwick and Uncles 1980; Shields et al. 2011).

A number of these studies use generalised hydrodynamic indexes or basic environmental descriptive categories, to relate the gradients of these regimes with patterns of ecological variation (Denny 2006). For example, wave exposure is often described as sheltered, moderately exposed, or exposed, with these categories used for ecological comparisons with species' community structural patterns (Nishihara and Terada 2010). The range of exposure within one individual category may however, be quite wide, and thus affect the interpretation

of ecological communities and the overall degree of ecological variation observed (Burrows et al. 2008).

A possible example of this is the intertidal rocky shorelines adjacent to offshore marine tidal race environments. At present, few in-depth tidal velocity reviews have been conducted within these types of environments, with ecological studies using broad tidal velocity categories, such as strong (>1.5-3 m s^{-1}) or very strong (> 3m s^{-1}), for ecological analysis and interpretation (Hiscock 1996; Shields et al. 2011). Yet, offshore marine tidal race environments undergo extremely strong hydrodynamic regimes on a daily basis, with tidal water speed potentially reaching up to 5.7 m s^{-1} (Myers and Bahaj 2005; Shields et al. 2009). These dynamic systems may therefore represent the highest part of the tidal velocity/ energetic properties of the descriptive category spectrum for study (Hiscock 1996; Shields et al. 2011). The tidal velocity categories however, may therefore miss the true tidal velocity description of the environment, and in turn the interpretation of ecological community patterns for study. In addition, the research strategy approach may limit the overall ecological interpretation of this specific environment type (or categorisation). Studies which use this approach frequently compare and describe the ecological patterns across the differing gradients (Wells et al. 2007). The underlying ecological and environmental mechanisms and processes which influence and drive the present ecological communities within this particular environment type may also be missed.

Therefore, these dynamic intertidal systems could be considered as useful new candidates for study, in terms of testing fundamental ecological theory, exploring patterns of ecological variation or the underlying mechanisms which drive them (Denny 2006). This could include testing the roles of ecological theories such as disturbance, succession, competition and recruitment (Dayton 1971; Connell and Slatyer 1977; Sousa 1979; Sousa 1984; Farrell 1989; Tilman 1990; Underwood et al. 2000; Chapman 2002). It could also involve detailed research of the biotic characteristics of species found within these environments, in terms of dispersal ability and life history traits, or, provide greater accuracy of the relationship between hydrodynamic properties and ecological component variation (Hart 1992; Steneck and Dethier 1994; Kim 1997; Menge et al. 2005; Burrows et al. 2008).

Currently, detailed ecological information for such investigations within these dynamic systems is sparse, as this environment is naturally uncommon across the global marine

landscape (Shields et al. 2011). Therefore, the aim of this chapter is to outline the ecological patterns within the intertidal regions of a marine tidal race environment. Here I use intertidal macroalgae and faunal species diversity, composition and colonisation, using descriptive observational studies and experimental based methods. This follows Okuda et al (2010), which suggest that using a range of taxonomies, measurable ecological parameters (including mature and immature community parameters) and research-method strategies can provide valuable information, for such investigations. The study aims to examine the ecological variation across differing spatial and temporal scales, within these dynamic systems. This is due to past studies outlining that ecological patterns can alter through time and space, which should be considered when examining ecological variation (Preston 1960; Underwood 2000; Menge et al. 2005). Overall, these parameters and research methods combined, allows this study to question what the patterns of ecological variation are within dynamic environments, such as the marine tidal race environments.

For this study, I examine ecological patterns across two intertidal rocky-shore bays, within a selected marine tidal race environment. This first provides site-specific information, which can be used to examine ecological variation 'within' this particular marine tidal race environment. The information can be used by other studies to enhance current ecological tidal velocity category descriptions of marine tidal race environments or further validate and examine ecological theory (described earlier). In addition, this selected marine environment is also currently under review for potential marine renewable energy developments, therefore, the information would be useful for future anthropogenic interaction investigation. The information could be incorporated into broader scaled marine monitoring programmes, which aim to examine ecological processes across different environmental gradients, geographical locations at long term temporal scales (Burrows 2012). As such, this may enable a more accurate ecological definition (over space and time) of marine tidal race environments, and in turn, the ecology the marine environment as a whole.

Here, I test the hypothesis that the intertidal rocky-shore ecological patterns (species diversity, assemblage composition and colonisation) within this environment vary across the measured spatial scales (two rocky-shore bays and shoreline height scales). I also test the hypothesis that these ecological patterns change over the measured temporal scales (seasonal and annual scales).

3.3. Methods

Study area

Two intertidal rocky-shore bays were selected adjacent to the marine tidal race environment known as the Alderney Race. The Alderney Race is located between the Channel Island of Alderney and the Cap de la Hague on the west coast of France. The two bays were located on the East coastline of Alderney, one locally named Longis Bay (49°43'300 N; 2°10'300 W) and another unnamed adjacent bay, which was given the name 'Control Bay' (49°43'300 N; 02°09'800 W), for the purpose of this study (Figure 3.1). This name was given due to the potential future development of Longis Bay for tidal energy development activities, with the Control Bay area used as a future control site for environmental impact assessment. The bays were divided by a concrete causeway and Victorian Fort building known as Fort Raz.

These bays were selected due to their similar substrate types, proximity to the tidal race environment and suitability for intertidal studies (other sections of this coastline are predominately cliffs). Substrate composition within these bays consist of similar mixtures of coarse sediments, cobbles and bedrock, with slightly larger proportions of sediments found within the middle of the Longis Bay (BGS 1989; BGS 1990; Admiralty Nautical Charts 2004; and personal observation during a 2008 pilot study). Longis Bay also contains a small pipe outlet from a small water treatment facility located at the North-West side of the bay in the upper shoreline. I deemed the outfall to have little nutrient enrichment impact on the overall survey area due to the small size of the overall facility, processing and nutrient output (ARE Ltd; personal communication).

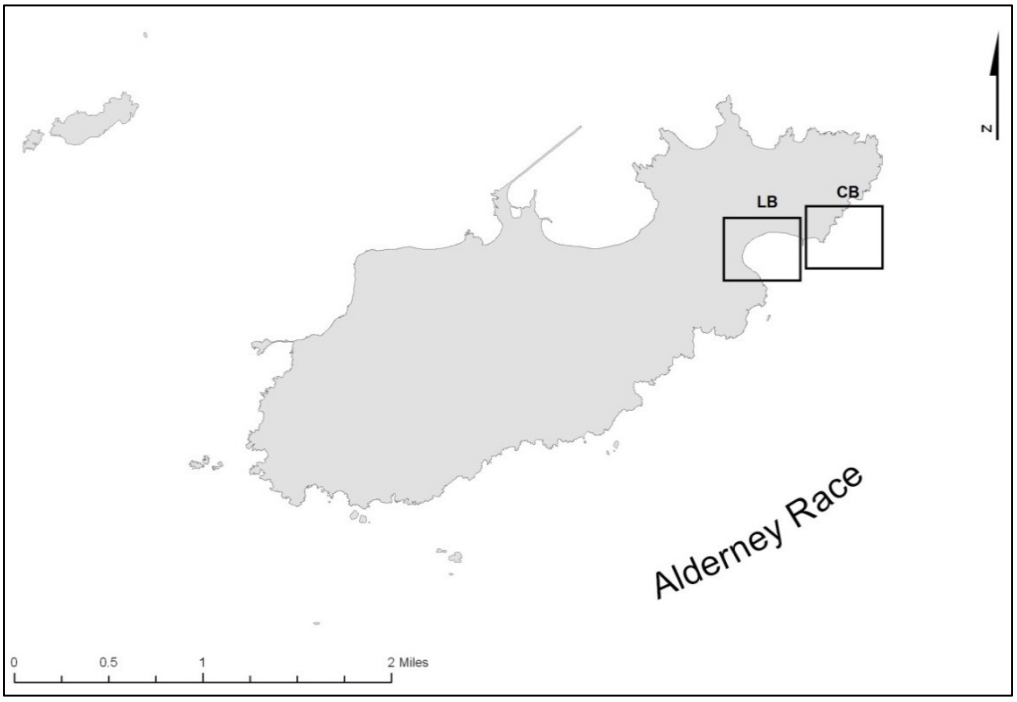


Figure 3.1. Location of the two intertidal bays selected for study, named Longis Bay (LB) and Control Bay (CB), from 2009 – 2011, situated on the East coastline of Alderney, Channel Islands. Intertidal bays are located along the intertidal coastline adjacent to the Alderney Race tidal race environment.

I split both bays into three separate shoreline height levels, and described them as upper, mid and lower shorelines (Skalski et al. 2001). These were based on their tidal shoreline heights, which ranged between the mean high water level of spring tides (MHWS) and the mean low water level of spring tides (MLWS). The position of these three shoreline heights were deduced from the mid-point between the MHWS and MLWS sections, based on Chart Datum (Admiralty Nautical Charts 2004), other chart tidal range information and a pilot study I completed in 2008 (ecological component comparisons between shoreline heights). The area of each shoreline height was approximately 500 metres in length and 150 meters height, within both bays. All field composition and colonisation sampling methods coincided with each selected month's spring tides, with site visits standardised to begin two hours prior to low tide (Nedwell et al. 2002; Horton and Murray 2006).

Intertidal community composition assessment

Intertidal community composition was assessed within both bays from 2009 to 2011, with temporal sampling periods every three months in February, May, August and November (seasonal) within each year, excluding November 2011.

I used a random stratified sampling technique, with four 100 metre (m) transect lines fixed perpendicular to the coastline within each shoreline height in both bays (Figure 3.2). The starting point of each transect line was positioned at the top of each shoreline height, with all transect lines placed in a Southerly direction (approximately 170° angle) and spaced approximately 100 m away from each other within both bays. These positions were based on the total length of the bays (approximately 500 m per bay) and the shoreline height area (approximately 500 m length x 150 m height), which were taken from Chart Datum information and trialed during the 2008 pilot study (Admiralty Nautical Charts 2004). Within each shoreline height I then further selected the position of the first transect approximately 100 metres from the West side/ edge of each bay (for example see Figure 3.2; labeled T1). All other transect positions within each shoreline height were then subsequently positioned accordingly. The start and end point of each transect was recorded using differential GPS (dGPS, with accuracy to 2.5m), with these same transect positions used throughout the study. Along each fixed transect, I assigned sampling station markers every 4 m, which were numbered 1 – 26 (from 0 to 100 m). Ten 0.5 m² gridded quadrats were then placed at randomly selected sampling stations along each transect and quadrat positions were randomised for each transect across each shoreline height, bay and temporal sampling period. The random station marker numbers (1-26) were generated in R software using random number generation techniques (R Development Core Team 2010). This method was repeated across each temporal sampling period.

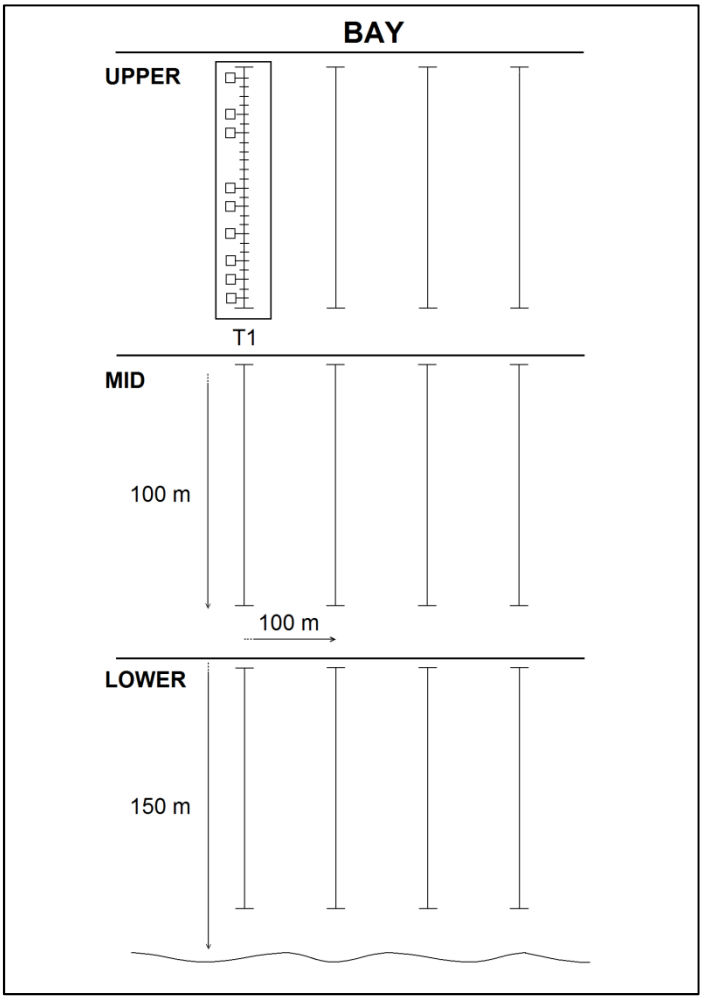


Figure 3.2. Schematic diagram of the 2009 – 2011 intertidal community compositional survey method conducted within one bay. The bay is split into upper, mid and lower shoreline heights, with four 100 metre transect lines running perpendicular to the shoreline within each shoreline height. On each transect (example shown as T1), ten 0.5 m^2 quadrats (shown as squares) were placed randomly along 26 pre-defined stations with a separation of 4m along each transect. This survey layout is replicated within each shoreline height, bay, and temporal sampling period.

Community composition was identified through visual species identification, estimation of macroalgae species percentage cover (%) and total faunal species (singleton/ non-colonial) abundance counts (n) within each quadrat (Nedwell et al. 2002). Each quadrat was photographed, with the quadrat position and sampling station number recorded using dGPS, for further species identification and review. Species were identified using authorative species keys and texts (Harvey 1849; Hayward and Ryland 1995; Hayward et al. 2001; Bunker et al. 2010; Fish and Fish 2011), with additional photographs and specimen collection

for further species identification from the field, particularly for problematic genera including *Porphyra*, *Ulva* and *Gibbula*.

Intertidal species colonisation study

Intertidal species colonisation was measured within the lower shoreline height, across both bays, from January 2009 to August 2011. This shoreline height was selected due to repeated loss of colonisation survey equipment from wave action at the other shoreline heights during the 2008 pilot study and initial 2009 surveys. Species colonisation was assessed by randomly distributing five experimental 0.5 m² clay tiles at the beginning of each survey year (January) within both bays (Figure 3.3). Tiles were fixed permanently to hard substratum (bedrock or large boulders), by drilling the back of the tiles into the substrate in two places (the top and bottom of each tile) with stainless steel screws. All tiles were placed at a 30 - 35° angle against the substratum, with the tile face orientated towards the sea. This was to minimise the risk of dislodgement from daily wave and sediment disturbance (personal observation).

Tiles were placed randomly within each bay according to the following scheme. A sampling region of 500m by 100m was identified in each bay, with the long axis running parallel to the shore. This was divided into five 100m by 100m sections along the shore. Each section was then divided into a grid of 50 rectangles, each of 20m by 10m, with the long axis again parallel to the shore. I then randomly selected one rectangle in each section for both bays every year for tile deployment locations. Random rectangle number (1-50 per section) was chosen using random number generation techniques in R (R Development Core Team 2010).The size of the rectangles was chosen on dGPS location quality for this region and because areas of this size are highly likely to contain hard substrate for tile attachment. At the end of each survey year (the start of December for 2009, 2010 and September for 2011), tiles were removed from both bays. This method was then repeated for the 2010 and 2011 survey years, with new tiles deployed each time, at different randomly selected rectangles locations. The position of each new tile was also recorded by dGPS (accuracy to approximately 2.5m).

To assess species colonisation, tiles were surveyed at monthly intervals across each year after the initial tile deployment within both bays. During each visit a 0.5 m^2 plastic gridded frame was laid over the top of each tile to aid in the estimation of macroalgae species percentage cover (%) and total faunal species abundance counts (n) were conducted in situ. Tiles were then photographed, with specimens taken where necessary for species identification (notably

Gibbula). Dense fronds of macroalgae attached to the tiles were also searched for hidden faunal and algal species.

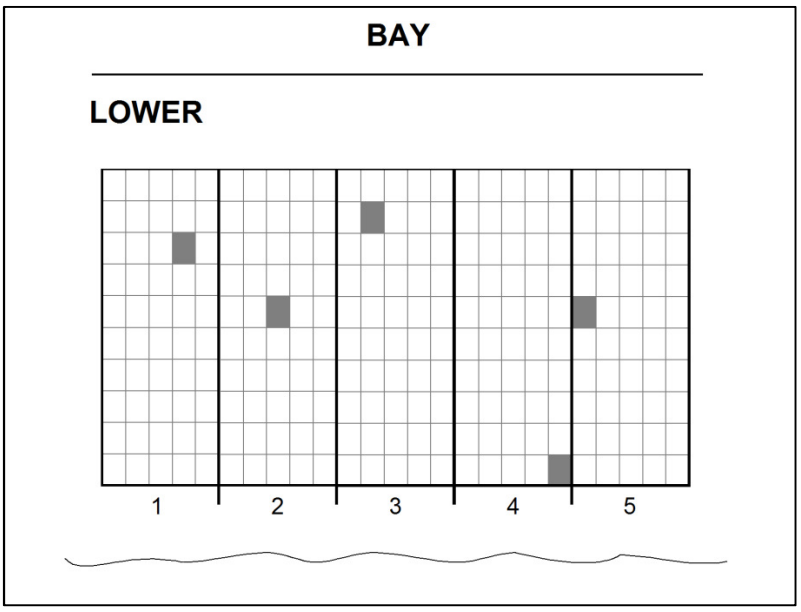


Figure 3.3. Schematic diagram of the 2009 – 2011 intertidal colonisation survey method conducted within the lower shoreline height of one bay. The bay is split into 5 gridded sub-sections (numbered 1-5), which comprise of the 50 numbered squares. One tile is randomly deployed within each sub-section, represented by grey squares. This survey layout is replicated within each bay and temporal annual sampling period with species colonisation measured at monthly intervals.

Data analysis

The R statistical software (R Development Core Team 2010) was used for all univariate analyses, with the Primer v6 software package used for exploratory multivariate routines (Clarke and Gorley 2006). I assessed intertidal community composition by averaging macroalgae percentage cover and faunal species abundance counts within each transect, giving four replicate average values for each bay (Longis Bay; Control Bay), within each shoreline height (upper; mid; lower) and across year (2009; 2010; 2011) and seasonal (February; May; August, November) temporal sampling periods, for a total of 88 data points for each shoreline height.

To examine species diversity differences between the two bays and temporal sampling periods for each shoreline height, I generated the diversity measures species richness (*S*),

Shannon-Wiener diversity (H', $\log_e$) and Pielou's evenness (J') using the Primer v6 DIVERSE routine. I then used generalised linear models (GLM) to compare these diversity measures as a function of bay and the annual, seasonal temporal sampling periods and their interactions, for each shoreline height. The species richness GLMs used a Poisson error distribution with a log link function and where overdispersion occurred, the quasi-poisson error distribution was used. The diversity descriptors, Shannon-Wiener and Pielou's evenness used the Gaussian error distribution. All models were fully factorial (including all interactions between the factor levels), with terms removed following model deletion methods (using ANOVA with appropriate F or Chi tests), if found not significant ($p > 0.05$) to produce minimum adequate models (MAM) (Crawley 2007). All minimal adequate models were then checked for non-normality and un-equal variance visually using Q-Q normality plots. I removed highly influential data points (outliers) within the macroalgae and faunal mid shoreline Pielou's evenness models. These data points were first examined from additional Cook's distance plots I generated for each of the models. I then removed these outliers (one at a time) only if they either created 5% or more decrease in unexplained variance when removed from the model (using model deletion techniques), or strongly altered the overall model when removed.

To assess species composition, I explored the macroalgae and faunal species datasets using multivariate analyses. All macroalgae and faunal datasets were square root transformed prior to these multivariate analyses, to moderately weight the contributions of common and rarer species for multivariate analyses (Clarke and Gorley 2006). I selected this transformation to undertake an overall moderate scope of these ecological datasets and from visual inspection of additional generated draftsman plots.

To examine if species composition differed between the two bays, years and seasonal temporal sampling periods within each shoreline height I used one-way analyses of similarities, with the Primer v6 ANOSIM routine. This routine tests groups of samples and compares the ranks of distance values within and between the selected groups. It gives a significance level, pairwise values and a global absolute value R which ranges from -1 (high differences within groups) to 1 (high differences between groups) and uses random permutations of the grouping factor to assess significance. This was run using the Bray-Curtis coefficient similarity resemblance (Legendre and Legendre 1998). I first ran this routine to compare macroalgae and faunal species composition between the two bays, for each

shoreline height separately. To compare the temporal patterns of macroalgae and faunal species composition, I then re-ran the routine, selected as a two-way ANOSIM analysis. This was assessed within each bay individually for each shoreline height, with year and season selected as factors.

To visualize the overall macroalgae and faunal species compositions between the two bays across all samples, I then generated non-multidimensional scaling ordination plots (nMDS: 2D) for each shoreline height. This was undertaken in the Primer v6 MDS routine, based on the generated Bray-Curtis resemblance matrices, with 50 algorithm cycle random restarts selected. This number of random restarts was chosen to increase the validation of selecting the best solution (lowest stress) of the generated nMDS plots, as the algorithm is iterative (Clarke and Warwick 2001). The derived stress values were then used to examine the goodness of fit for each nMDS ordination plot generated, for data summary interpretation means (where < 0.05: excellent; < 0.1: good; > 0.3 poor nMDS ordination plot representation of the datasets) (Clarke and Warwick 2001).

To assess macroalgae species colonisation, I averaged the percentage cover across the five tiles within each bay (LB; CB) for each month (January – November) and each year (2009; 2010; 2011). To assess faunal species I used the total number of faunal species individuals (n) within each bay, for each month and year temporal sampling period. I then used GLMs to compare macroalgae and faunal species diversity and evenness between the two bays and three years (total n = 60). This included calculating species richness (*S*), Shannon-Wiener diversity (H', $\log_e$) and Pielou's evenness (J') for each month, within both bays and years using the Primer v6 DIVERSE routine. I used the GLMs to compare these diversity measures as a function of bay, annual temporal scales and their order interactions, to produce MAMs (for GLM model deletion methods see above). All MAM models were checked for unequal variance and non-normality using Q-Q normality plots.

For multivariate analyses, I first square root transformed the macroalgae species dataset, with faunal species dataset log transformed (Clarke and Gorley 2006). I then used the Primer v6 one-way ANOSIM routine to explore macroalgae and faunal species compositional differences between the two bays and three years (see ANOSIM method description above). The Gower coefficient resemblance measure was selected for these multivariate analyses, due to its usefulness for immature communities where small samples occur (Legendre and

Legendre 1998). The ANOSIM routine was first run to assess macroalgae and faunal species composition between the two bays across the entire tile colonisation datasets. To examine the temporal compositional differences between the three years, I then re-ran this routine within each bay dataset separately.

To outline which macroalgae and faunal species contributed to the compositional differences shown by the ANOSIM results, I used the Primer v6 similarity percentage contribution SIMPER routine. This routine is based on the Bray-Curtis similarity resemblance measure and determines the percentage contributions (%) to the average Bray-Curtis dissimilarity between pairs of groups (Clarke and Gorley 2006). I first ran this routine as a one-way analysis across the entire macroalgae and faunal colonisation datasets; to identify the species percentage contributions between the two bays. I then ran this routine within each bay dataset separately; to identify the species percentage contributions between the three years. For all SIMPER analyses, I selected a 100 % cut off level to show all species percentage contributions, due to the small number of species recorded during this study.

3.4. Results

a) Composition analyses

A total number of 47 macroalgae and 25 faunal species were recorded across Longis Bay and Control Bay during the 2009 – 2011 intertidal community composition study (see Appendix 1 Table A.1 for a full species and taxonomic list). Macroalgae species included 7 Chlorophyta, 14 Phaeophyta, 25 Rhodophyta and also 1 Angiospermae species. Intertidal faunal species included 1 Annelida, 1 Arthropoda, 1 Cnidaria, 3 Crustacea, 1 Echinodermata and 18 Mollusca species.

Macroalgae species diversity showed considerable variation between the measured spatial and temporal scales and selected diversity measures (Figure 3.4; Table 3.1). Faunal species diversity measures showed more consistent spatial and temporal variation (Figure 3.5; Table 3.2). Overall, macroalgae species composition generally altered at the spatial scale, with faunal species composition variable at both the spatial and temporal scales (Tables 3.3-3.4; Figure 3.6).

Macroalgae diversity

My results showed that macroalgae species diversity varied between the two bays across the three shoreline heights, years and seasons (Figure 3.4).

In the upper shoreline, I observed significantly larger species richness (*S*) and Shannon-Wiener (H') diversity within the Control Bay, with differences shown by the GLM results (Table 3.1). The two bays showed a similar distribution of species abundances, outlined by the Pielou's evenness measure (J'). I found that within this shoreline height, macroalgae species richness significantly declined across the three years. The Shannon-Wiener and Pielou's evenness measures however, showed similar annual differences. At the seasonal scale, these diversity measures outlined similar estimates of macroalgae diversity overall.

Within the mid shoreline height, I also found greater macroalgae species richness and Shannon-Wiener diversity within the Control Bay, with differences shown by the GLM results. In addition, Pielou's evenness outlined comparable estimates between these two bays. Macroalgae diversity and evenness showed considerable differences season and year and their interaction (Table 3.1). I also found with all diversity measures that 2010 comprised of the largest diversity and evenness estimates at the annual scale and August comprising with the fewest number of species at the seasonal scale.

In the lower shoreline, I found that all diversity measures outlined similar macroalgae diversity patterns between the two bays and temporal scales, shown by the GLM results. I also however, found significant seasonal differences of Pielou's evenness, with the smallest differences in abundance found during the May sampling periods.

Faunal diversity

Faunal species diversity patterns also varied across the measured spatial and temporal scales (Figure 3.5). In the upper shoreline, all diversity measures showed significantly larger faunal diversity within the Control Bay, shown by the GLM results (Table 3.2). I also however, found large Pielou's evenness estimates within Longis Bay. Within this shoreline, all diversity measures showed similar faunal diversity estimates across the annual and seasonal sampling periods.

Within the mid shoreline, the GLM species richness and Shannon-Wiener diversity results identified that faunal diversity was significantly larger within the Control Bay. These results however, also highlighted that Pielou's evenness was substantially larger within the Longis Bay. Across the three years, species richness and Shannon-Wiener measures both showed significant declines of faunal diversity, whilst Pielou's evenness showed significantly larger estimates during 2010. Overall, I found all diversity measures outlined similar faunal diversity across the seasonal sampling periods.

The lower shoreline was also represented by larger species richness and Shannon-Wiener diversity estimates within the Control Bay, and similar Pielou's evenness between the two bays. At the temporal annual scale, species richness showed a significant decline across the three years, whilst Shannon-Wiener and Pielou's evenness showed similar temporal diversity. Overall, these diversity measures showed similar diversity estimates between the seasonal sampling periods.

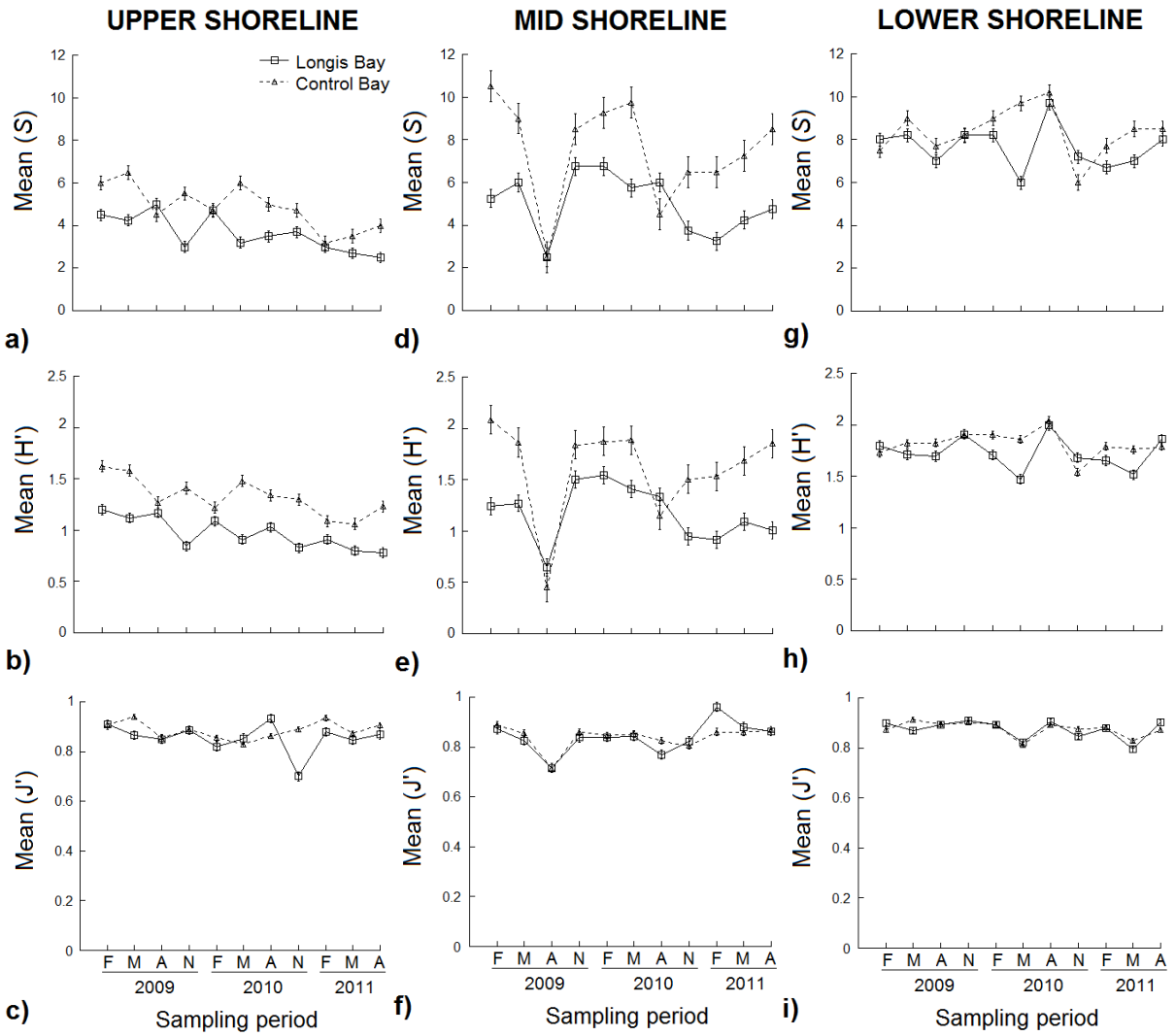


Figure 3.4. Mean intertidal macroalgae species richness (*S*), Shannon-Wiener (H', $\log_e$) and Pielou's evenness (J') diversity measures per seasonal sampling period for each shoreline height (upper a-c, mid d-f, and lower g-i) across Longis Bay and the Control Bay, from 2009 – 2011, Alderney. Solid line represents Longis Bay, dotted line represents the Control Bay (± 1 SEM). Letters represent the seasonal sampling periods (F = February, M = May, A = August, N = November).

Table 3.1. Generalised linear minimal adequate models (MAM) for intertidal macroalgae species diversity measures within the upper, mid and lower shoreline heights, from 2009 - 2011, Alderney. Species diversity measures include species richness (*S*), Shannon-Wiener (H', $\log_e$) and Pielou's evenness (J') diversity. F represents the overall F model value; P represents the model P value; DF represents the degrees of freedom; n.s. represents no significant variables (n = 88).

Shoreline height	Model explanatory variables	F value	P	DF
Upper	**Species richness (*S*)**			
	Intercept			
	Bay	5.18	0.02	86
	Year	4.21	0.01	85
	Shannon-Wiener diversity (H')			
	Intercept			
	Bay	9.78	0.002	86
	Pielou's evenness (J')	n.s		
Mid	**Species richness (*S*)**			
	Intercept			
	Bay	14.825	< 0.001	86
	Year:Season	3.81	0.003	77
	Shannon-Wiener diversity (H')			
	Intercept			
	Bay	13.52	< 0.001	86
	Season	3.15	0.02	85
	Year:Season	3.52	0.006	77
	Pielou's evenness (J')			
	Intercept			
	Year	4.47	0.01	76
	Season	4.41	0.006	75
Lower	**Species richness (*S*)**	n.s		
	Shannon-Wiener diversity (H')	n.s		
	Pielou's evenness (J')			
	Intercept			
	Season	2.81	0.04	84

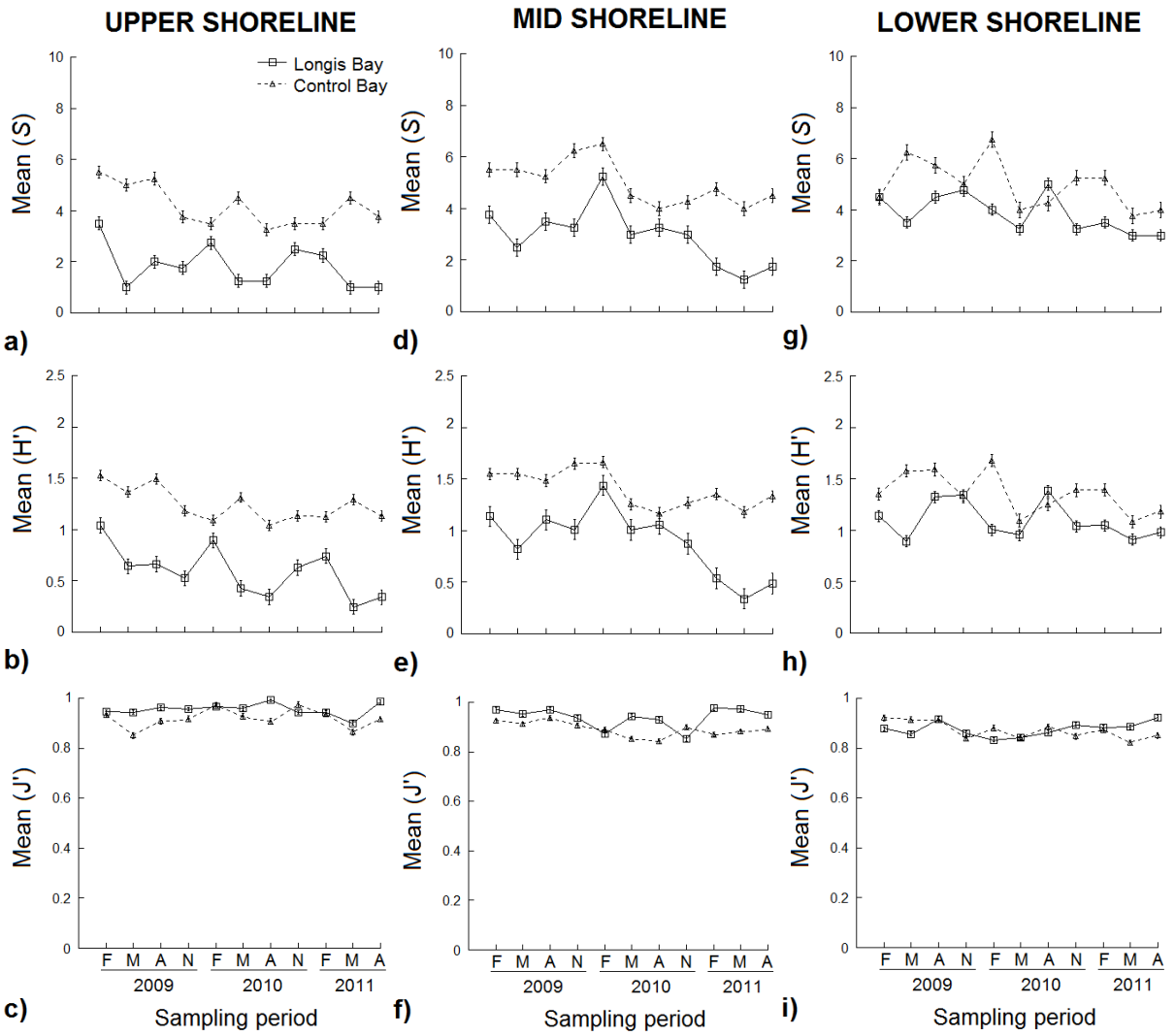


Figure 3.5. Mean intertidal faunal species richness (*S*), Shannon-Wiener (H', $\log_e$) and Pielou's evenness (J') diversity measures per seasonal sampling period for each shoreline height (upper a-c, mid d-e, and lower g-i) across Longis Bay and the Control Bay, from 2009 – 2011, Alderney. Solid line represents Longis Bay, dotted line represents the control (± 1 SEM). Letters represent the seasonal sampling periods (F = February, M = May, A = August, N = November).

Table 3.2. Generalised linear minimal adequate models (MAM) for intertidal faunal species diversity measures within the upper, mid and lower shoreline heights, from 2009 - 2011, Alderney. Species diversity measures include species richness (*S*), Shannon-Wiener (H', $\log_e$) and Pielou's evenness (J') diversity. F represents the overall F model value; P represents the model P value; DF represents the degrees of freedom; n.s. represents no significant variables (n = 88).

Shoreline Height	Model explanatory variables	F value	P	DF
Upper	**Species richness (*S*)**			
	Intercept			
	Bay	30.51	< 0.001	86
	Shannon-Wiener diversity (H')			
	Intercept			
	Bay	40.54	< 0.001	86
	Pielou's evenness (J')			
	Intercept			
	Bay	6.46	0.01	86
Mid	**Species richness (*S*)**			
	Intercept			
	Bay	24.00	< 0.001	86
	Year	4.20	0.01	85
	Shannon-Wiener diversity (H')			
	Intercept			
	Bay	30.44	< 0.001	86
	Year	6.84	0.01	85
	Bay:Year	3.32	0.04	82
	Pielou's evenness (J')			
	Intercept			
	Bay	13.89	< 0.001	74
	Year	8.10	< 0.001	73
Lower	**Species richness (*S*)**			
	Intercept			
	Bay	10.59	0.001	86
	Year	3.37	0.03	85
	Shannon-Wiener diversity (H')			
	Intercept			
	Bay	9.81	0.002	86
	Pielou's evenness (J')	n.s		

Species composition

Macroalgae

The ANOSIM and nMDS results showed that macroalgae species composition differed between the two bays, across the three shoreline heights (Table 3.3; Figure 3.6a-c). Additional SIMPER analyses revealed this was due to Longis Bay comprising of *Ascophyllum nodosum* within the upper and mid shorelines and also larger estimates of *Fucus serratus* recorded within the lower shoreline (see Appendix 3 Tables A.3.1-3.3 for SIMPER analyses).

From these results, I found that the two bays largely comprised of similar species composition across the temporal annual and seasonal scales, within each shoreline height. I also found however, temporal compositional differences within the mid and lower shorelines of the Control Bay, shown by the ANOSIM pairwise results.

Within the mid shoreline, species composition differed in 2009, which was due to the added presence of *Corallina officinalis* compared to 2010 and 2011, outlined by the additional SIMPER results (Appendix 3 Table A.3.2). Seasonal differences in this shoreline were attributed to the fluctuating dominance of fucoid species such as *Fucus vesiculosus* and the presence of chlorophyta species including *Ulva lactuca*.

Within the lower shoreline, species composition differed between the February and August seasonal sampling periods. The additional SIMPER results outlined this was due to the increased presence of species including *Ulva intestinalis*, *F. serratus* and *Himanthalia elongata* within the August season (Appendix 3 Table A.3.3).

Fauna

The composition of faunal species between the two bays within each shoreline height significantly differed, which was identified by the ANOSIM and nMDS results (Table 3.4; Figure 3.6d-f). The additional SIMPER results outlined that this was due to the larger proportions of *Patella vulgata* recorded within Longis Bay and *Osilinus lineatus* within the Control Bay; across all shoreline heights (see Appendix 3 Tables A.3.4-3.6).

These results also outlined that the composition of species within both bays differed across the temporal annual and seasonal scales.

Longis Bay within the mid shoreline height showed some annual species compositional differences, specifically during 2010, outlined by the ANOSIM pairwise results. The additional SIMPER results showed this was due to the large proportion of *Gibbula cineraria* recorded in this year (Appendix 3 Table A.3.5).

Within the lower shoreline height of this bay, species composition changed annually. I found this was due to the due to the fluctuating presence of molluscan species such as *O. lineatus* (Appendix 3 Table A.3.6). Overall, these results show weak compositional differences between the seasonal sampling periods.

Within the Control Bay, the ANOSIM and additional SIMPER results (Appendix 3 Tables A.3.4-3.6) showed significant species compositional differences across all the annual and seasonal scales, within the three shoreline heights overall.

Table 3.3. ANOSIM results for intertidal macroalgae species composition at each shore height at three levels: a) differences between bays, b) differences within bays across years and c) differences within bays across seasons. Global R (R), pairwise values and significance levels (*P*) are based on one-way ANOSIM routines between bays (one-way factor: bay) and within each bay (two-way factors: bays plus year or season).

	Global values							
a) Bay	**R**	***P***	**Pairwise R values**					
Shoreline								
Upper	0.266	0.001						
Mid	0.243	0.001						
Lower	0.291	0.001						
			2009 - 2010	**2009 - 2011**	**2010 - 2011**			
b) Year								
Shoreline								
Upper								
Longis Bay	-0.168	0.992	-0.156	-0.146	-0.188			
Control Bay	-0.014	0.589	-0.052	0.056	-0.038			
Mid								
Longis Bay	-0.164	0.998	-0.125	-0.215	-0.17			
Control Bay	0.226	0.003	0.182	0.42	0.038			
Lower								
Longis Bay	0.117	0.10	0.138	0.184	0.045			
Control Bay	0.081	0.143	0.141	0.16	-0.049			
			Feb - May	**Feb - Aug**	**Feb - Nov**	**May - Aug**	**May - Nov**	**Aug - Nov**
c) Season								
Shoreline								
Upper								
Longis Bay	-0.165	0.996	-0.177	-0.139	-0.219	-0.146	-0.198	-0.13
Control Bay	-0.032	0.702	-0.108	0.01	-0.026	0	0.005	-0.01
Mid								
Longis Bay	-0.151	0.994	-0.146	-0.17	-0.109	-0.236	-0.141	-0.047
Control Bay	0.324	0.001	0.201	0.60	0.182	0.271	0.266	0.52
Lower								
Longis Bay	0.083	0.102	0.174	0.205	0.031	0.049	0.047	-0.03
Control Bay	0.131	0.03	0.069	0.30	0.01	0.038	0.083	0.167

Table 3.4. ANOSIM results for intertidal faunal species composition at each shore height at three levels: a) differences between bays, b) differences within bays across years and c) differences within bays across seasons. Global R (R), pairwise values and significance levels (*P*) are based on ANOSIM routines between bays (one-way factor: bay) and within each bay (two-way factors: bays plus year or season) for each shoreline height.

	Global values							
a) Bay	**R**	***P***	**Pairwise R values**					
Shoreline								
Upper	0.526	0.001						
Mid	0.421	0.001						
Lower	0.298	0.001						
			2009 - 2010	**2009 - 2011**	**2010 - 2011**			
b) Year								
Shoreline								
Upper								
Longis Bay	-0.028	0.601	-0.023	-0.021	-0.003			
Control Bay	0.407	0.001	0.388	0.54	0.344			
Mid								
Longis Bay	0.217	0.009	0.19	-0.007	0.49			
Control Bay	0.464	0.001	0.42	0.69	0.25			
Lower								
Longis Bay	0.296	0.001	0.28	0.20	0.36			
Control Bay	0.663	0.001	0.87	0.87	0.28			
			Feb - May	**Feb - Aug**	**Feb - Nov**	**May - Aug**	**May - Nov**	**Aug - Nov**
c) Season								
Shoreline								
Upper								
Longis Bay	-0.014	0.526	0.01	0.049	-0.068	-0.181	0.031	0.112
Control Bay	0.296	0.001	0.517	0.41	0.344	0.028	0.36	0.156
Mid								
Longis Bay	0.028	0.311	0.038	0.012	0.161	-0.069	0.115	-0.036
Control Bay	0.18	0.009	0.069	0.049	0.51	0.15	0.266	0.349
Lower								
Longis Bay	0.119	0.028	-0.028	0.097	0.26	0.076	0.37	0.12
Control Bay	0.277	0.001	0.236	0.125	0.469	0.142	0.56	0.44

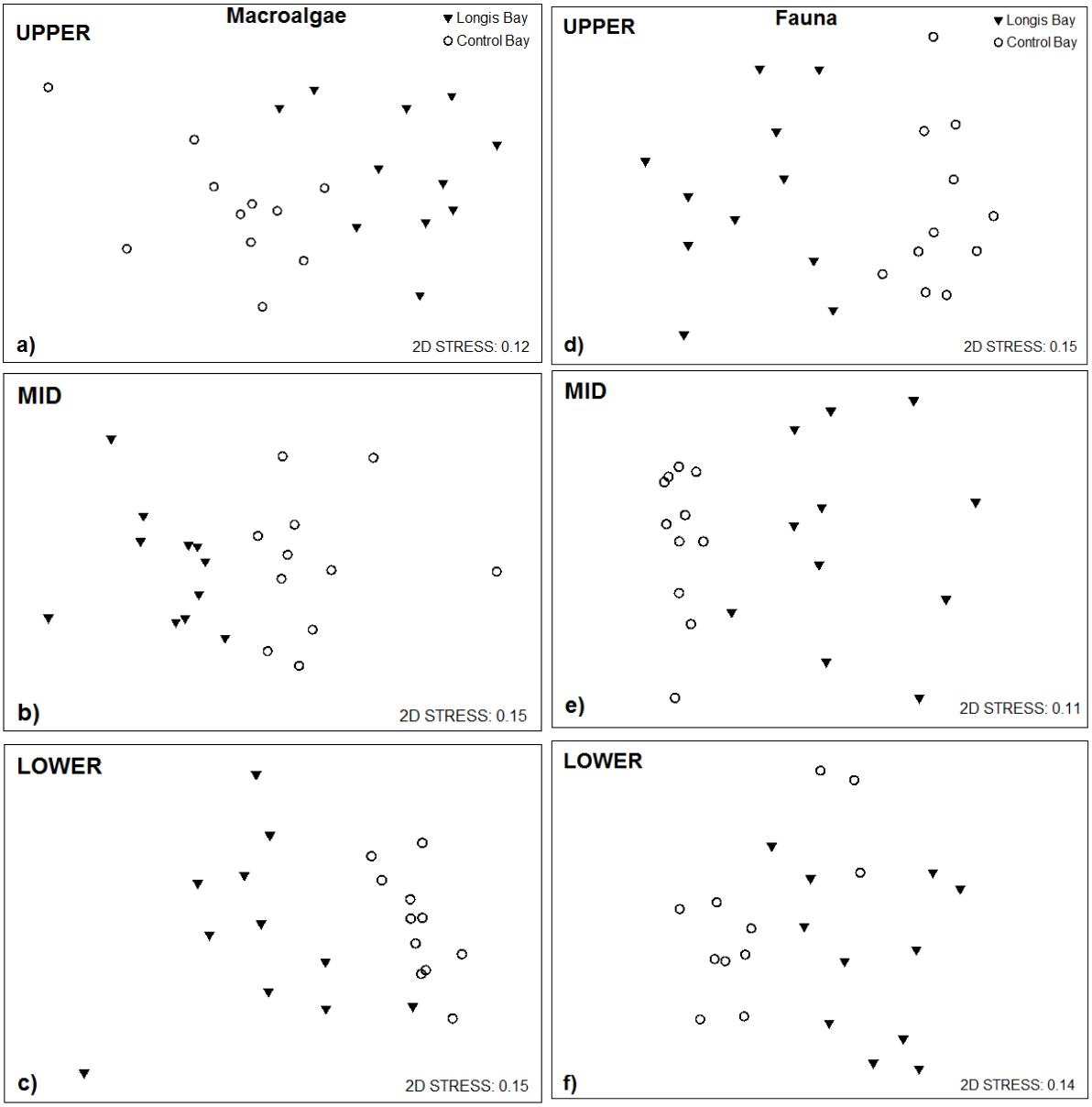


Figure 3.6. Non-metric 2-D multidimensional scaling ordination plots of intertidal macroalgae species composition for upper a), mid b), and lower c), shoreline heights and, faunal species composition for upper d), mid e), and lower f), shoreline heights, from 2009 - 2011, Alderney. Triangles represent Longis Bay; circles represent the Control Bay. Ordination plots are based on mean macroalgae percentage cover and mean faunal abundance data for each season within each shoreline height. Stress values are given, where < 0.05: excellent; < 0.1: good; > 0.3 poor representation.

b) Colonisation study

The colonisation study recorded a total number of 11 intertidal macroalgae and 15 faunal species colonising the experimental tiles across the two bays. Macroalgae species included 5 Chlorophyta, 2 Phaeophyta and 4 Rhodophyta species, with faunal species represented by 1 Cnidaria, 1 Crustacea, 1 Echinodermata and 12 Mollusca species.

Overall, colonising macroalgae species showed different annual patterns of growth, diversity estimates and species composition between the two bays (Figure 3.7; Tables 3.5-3.7). Faunal species generally showed similar patterns of abundance, diversity estimates and species composition between the two bays, which also changed over time, annually (Figure 3.8; Tables 3.5-3.7).

Macroalgae

The pattern of macroalgae cover altered between the years within both bays (Figure 3.7 a-c). Longis Bay showed substantial macroalgae growth during each year across the colonisation tiles, particularly after the May sampling period. I observed this pattern within the Control Bay during 2009; however, during 2010 and 2011, macroalgae cover showed reduced growth, which fluctuated slightly across the sampling months.

The GLM diversity results show that macroalgae species richness (*S*), Shannon-Wiener (H') and Pielou's evenness (J') were all significantly larger within Longis Bay (Table 3.5). These results also showed similar measures of species diversity and evenness between the three years for both bays overall (Figure 3.7 d-f).

Fauna

Faunal species showed similar patterns of abundance within both bays, which differed between the three years (Figure 3.8 a-c). In 2009, species abundance increased dramatically during the September sampling period within both bays. However, 2010 and 2011, showed increased abundance from April and March respectively.

Overall, I found comparable estimates of species diversity and evenness between the two bays (Table 3.5). Across the three years, both bays also showed similar species diversity and evenness estimates (Figure 3.8 d–f).

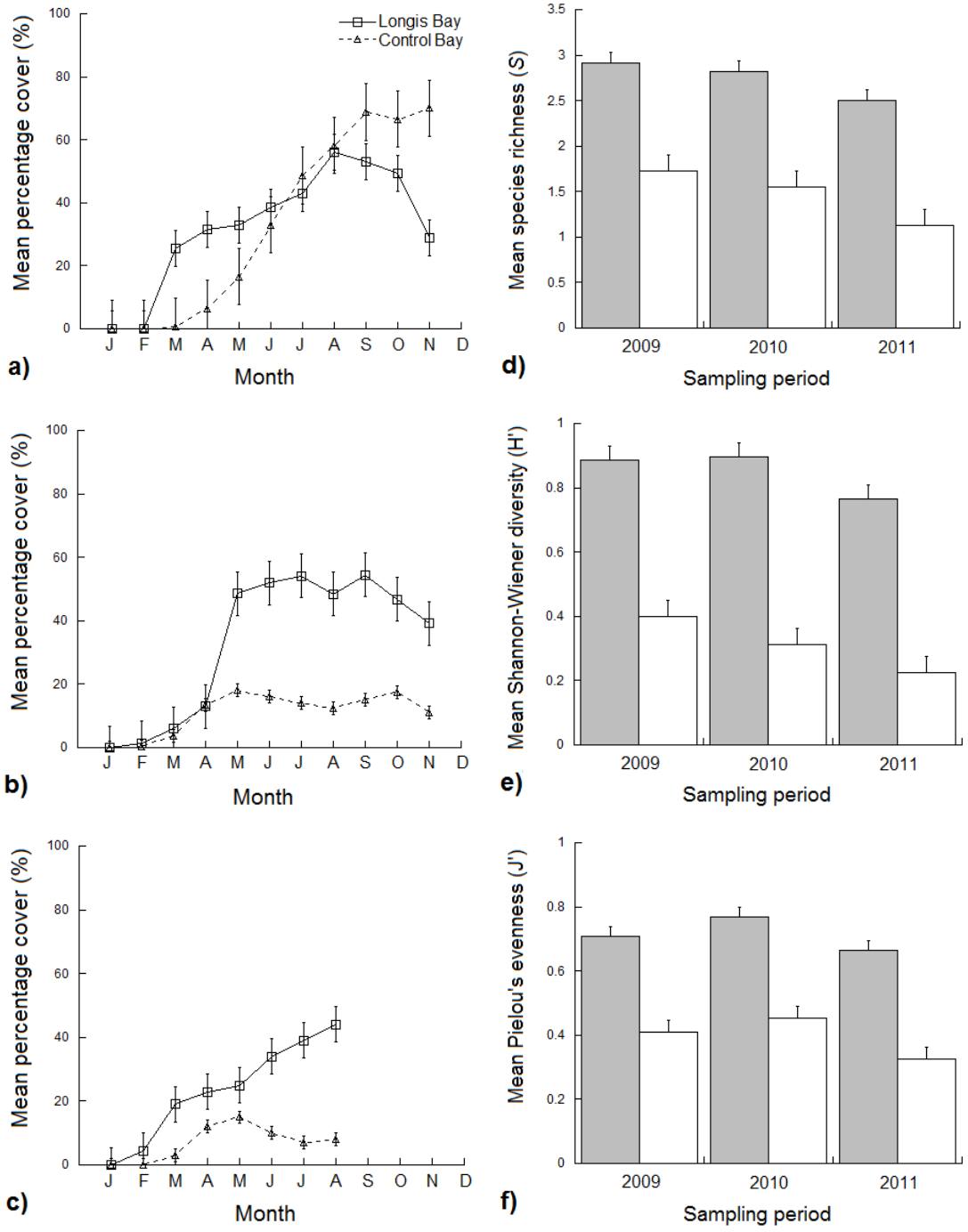


Figure 3.7. a-c) Within year patterns in percentage cover and d-f) between year differences in species diversity measures for intertidal macroalgae colonising the experimental tiles. Monthly mean percentage cover within each bay (see legend in panel a) are estimated across each tile (n = 5) for a) 2009, b) 2010 and c) 2011. Mean diversity measures within years for are shown Longis Bay (grey bars) and Control Bay (white bars) for d) species richness (*S*), e) Shannon-Wiener (H', $\log_e$) and f) Pielou's evenness (J'). All error bars are ± 1 SEM.

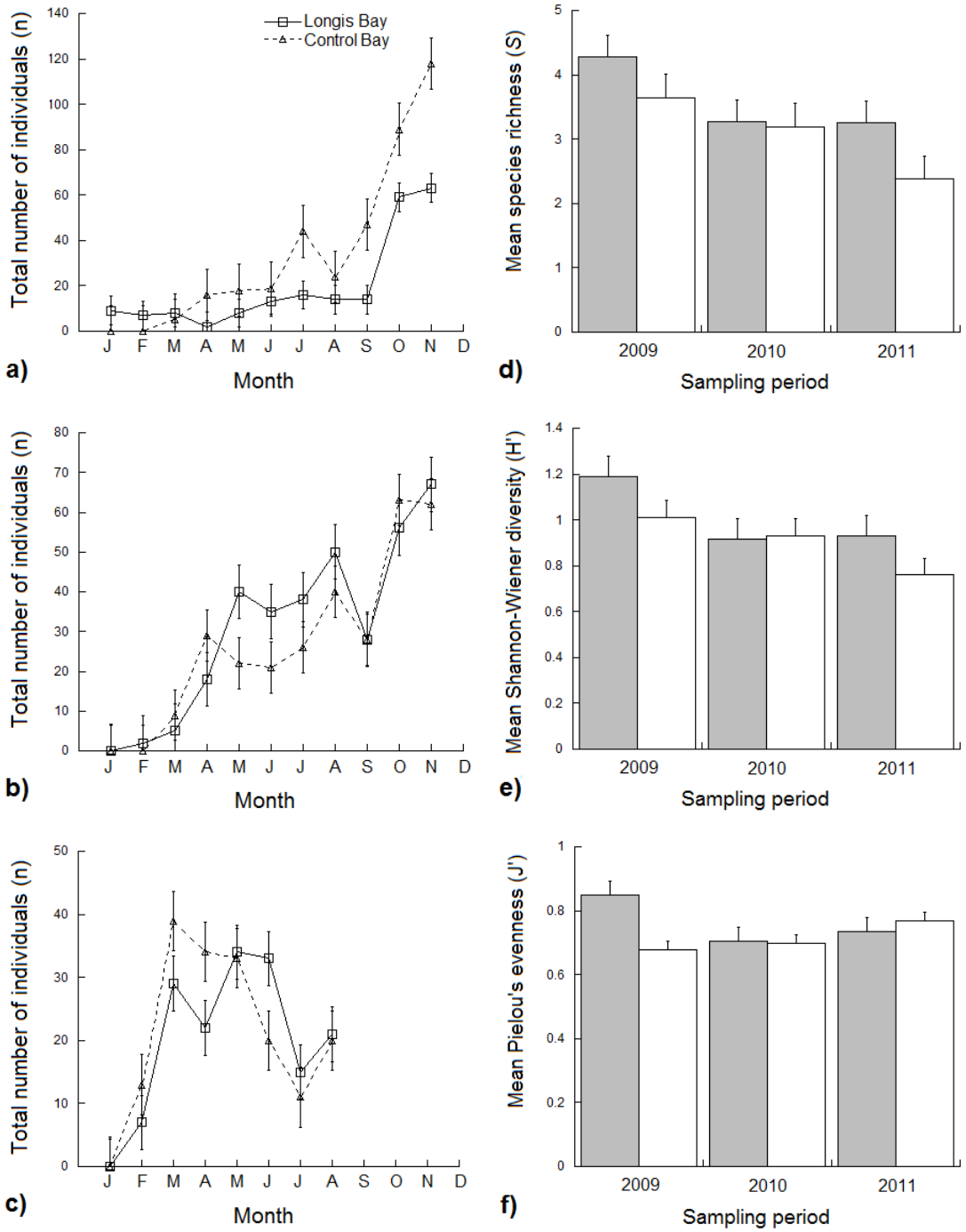


Figure 3.8. a-c) Within year patterns in abundance counts and d-f) between year differences in species diversity measures for intertidal faunal species colonising the experimental tiles. Monthly total abundance within each bay (see legend in panel a) are estimated across each tile (n = 5) for a) 2009, b) 2010 and c) 2011. Mean diversity measures within years for are shown Longis Bay (grey bars) and Control Bay (white bars) for d) species richness (*S*), e) Shannon-Wiener (H', $\log_e$) and f) Pielou's evenness (J'). All error bars are ± 1 SEM.

Table 3.5. Generalised linear minimal adequate models (MAM) for intertidal macroalgae species and faunal species diversity measures within the colonisation study, from 2009 - 2011, Alderney. Species diversity measures include species richness (*S*), Shannon-Wiener (H', $\log_e$) and Pielou's evenness (J') diversity. F represents the overall F model value; P represents the model P value; DF represents the degrees of freedom; n.s. represents no significant variables (n = 60).

Species	Model explanatory variables	F value	P	DF
Macroalgae	**Species richness (*S*)**			
	Bay	11.45	< 0.001	58
	Shannon-Wiener diversity (H')			
	Bay	24.73	< 0.001	58
	Pielou's evenness (J')			
	Bay	9.68	0.002	58
Fauna	**Species richness (*S*)**	n.s		
	Shannon-Wiener diversity (H')	n.s		
	Pielou's evenness (J')	n.s		

Species composition

Macroalgae

The composition of macroalgae species colonising the experimental tiles significantly differed between the two bays, shown by the ANOSIM results (Table 3.6). From the SIMPER results, I found that this was due to the presence of *Dilsea carnosa* (Rhodophyta) within Longis Bay (Table 3.7). These results also showed that within each bay, species composition was similar throughout the three years, with a small number of species contributing to the overall composition. In Longis Bay, three dominant species were recorded colonising the tiles, whilst only two were recorded in the Control Bay.

Fauna

The ANOSIM and SIMPER results identified that faunal composition across the tiles between both bays were highly similar (Table 3.6; Table 3.7). These results also showed that the Control Bay showed comparable species composition between the three years. Within Longis Bay however, I also found that species composition altered annually. The SIMPER results identified that these annual compositional differences were attributed to the

fluctuating dominance of the molluscs, *Osilinus lineatus*, *Patella vulgata* and *Gibbula cineraria.*

Table 3.6. ANOSIM colonisation study results of intertidal macroalgae and faunal species for Longis Bay and the Control Bay, across 2009 – 2011, within the lower shoreline height, Alderney. Global R (R), pairwise values and significance levels (*P*) are based on separate one-way ANOSIM routines of macroalgae and faunal species datasets; between bays (factor: bay) and within each bay (factor: year).

	Global values		**Pairwise R values**		
	R	***P***	**2009 - 2010**	**2009 - 2011**	**2010 - 2011**
Macroalgae					
Bay	0.214	0.001			
Year					
Longis Bay	-0.02	0.63	-0.04	-0.03	-0.05
Control Bay	0.07	0.07	0.09	0.13	-0.01
Fauna					
Bay	0.036	0.05			
Year					
Longis Bay	0.157	0.008	0.12	0.17	0.19
Control Bay	0.039	0.18	0.06	0.03	-0.02

Table 3.7. SIMPER percentage contribution (%) colonisation study results of intertidal macroalgae and faunal species within Longis Bay (LB) and the Control Bay (CB), from 2009 – 2011, within the lower shoreline height, Alderney. Average similarity represents the similarity of the samples within each factor level, with bold representing ≥ 20 % proportion contributions.

	Bay		**Year**					
	LB	**CB**	**LB**			**CB**		
			2009	**2010**	**2011**	**2009**	**2010**	**2011**
Macroalgae								
Chlorophyta								
Cladophora rupestris	0.37		3.20					
Ulva intestinalis	**50.24**	**56.72**	**49.20**	**41.75**	**62.90**	**22.66**	**40.11**	**97.28**
Ulva linza		0.71				5.43		
Phaeophyta								
Fucus vesiculosus	**29.72**	**41.79**	**31.49**	**34.88**	18.77	**71.61**	**57.58**	2.72
Rhodophyta								
Dilsea carnosa	19.16		16.12	**22.22**	17.47			
Lithophyllum spp	0.52			1.15	0.86			
Porphyra umbilicalis		0.78				0.30	2.31	
Average similarity	47.38	28.71	43.01	50.02	48.13	29.28	33.31	36.74
Fauna								
Cnidaria								
Actinia equina	0.29		1.29					
Echinodermata								
Asterina gibbosa	0.03							
Mollusca								
Gibbula cineraria	12.99	6.61	12.54	**30.42**		16.18	10.18	
Gibbula pennanti	0.57	1.35	0.24	1.39		5.54	0.68	
Gibbula umbilicalis	5.20	7.24	2.17	3.46	5.00	3.03	7.80	7.71
Littorina littorea	2.07	0.07		2.04	3.21	0.23		
Littorina fabalis	0.37				3.38			
Littorina obtusata	0.36	0.30	0.96			2.26		
Littorina saxatilis	2.02	1.35	13.97			13.43		
Osilinus lineatus	**45.35**	**47.13**	**46.02**	13.18	**58.47**	**51.99**	**32.41**	**44.09**
Patella vulgata	**28.06**	**35.95**	4.20	**49.51**	**29.94**	7.35	**48.93**	**48.20**
Patella depressa	2.69		18.60					
Average similarity	28.70	35.96	32.75	34.47	48.25	28.35	40.37	53.71

3.5. Discussion

My results overall show that ecological components of the intertidal regions of this marine tidal race environment show spatial and temporal patterns of variation. This was observed across varying scales, in terms of species diversity, composition and colonisation of both macroalgae and faunal species.

Species community diversity

Spatial ecological variation within the marine environment is described as a function of intrinsic and extrinsic factors, which include dispersal, species interactions or biological and physical disturbance (Menge et al. 2005; Balata et al. 2006). Intertidal shoreline heights are also subjected to differing gradients of extrinsic factors including tidal flow conditions, or thermal and desiccation stress, which are known to increase towards the upper regions of the shoreline (Menge et al. 2005). The aspect of the coastline, distance to offshore open water, wave fetch, general shoreline profile, gradient and inclination can also result in different levels of wave attenuation across the shoreline. Exposed shorelines are known to comprise of a mosaic of macroalgae and faunal species, with highly exposed or more vertical profile shorelines dominated by barnacle and mussel complexes (Wells et al. 2007). Menge et al (2005), also describe how low (sloping), sheltered shorelines can be covered by a range of multi-specific macroalgae species. Therefore, for large shorelines, such as the two bays studied here, the three shoreline heights may therefore comprise of different profiles and inclinations. As such macroalgae and faunal species diversity may be heavily influenced, which is represented in the variation I found between the two bays and the shoreline heights during this study.

Ecological temporal variation at the annual scale is often related to a range of physical disturbance regimes, such as increased hydrodynamic conditions during the winter periods (Guerra-García et al. 2011). These regimes can vary in their occurrence and intensity, with influence also depending upon the susceptibility/ life stage of the assemblage to the disturbance type (Sousa 1984; Menge et al. 2005). Seasonal variation is often linked to periodic biological interactions such as species recruitment, reproduction or foraging activities (Wernberg and Goldberg 2008; Noël et al. 2009; Spatharis et al. 2011). Overall, my study showed that macroalgae and faunal species diversity temporal variation within this marine tidal race environment altered differently across the three shoreline heights (specifically within the mid shoreline height). This suggests that within these dynamic

environments, there is a potential graduated intensity of interacting intrinsic and extrinsic factors across the shorelines, which influence intertidal species diversity at differing temporal scales.

Within the lower shorelines, I also found that species diversity, particularly macroalgae species diversity, was generally constant throughout the timescale of this study. This may indicate the ability of species to either recover or persist in the face of the intense wave action found within this level (Menge et al. 2005; Balata et al. 2006; Shields et al. 2011). Menge et al (2005) suggest that recovery from such disturbance regimes is a reflection of the local processes influencing the colonisation, growth and diversity of species assemblages. In terms of recoverability, species turnover may therefore occur at faster (such as weeks) or slower rates (such as years) rather than the measured sampling scales used in this study (Wernberg and Goldberg 2008). The true pattern of the underlying temporal patterns may be missed in this shoreline and environment type as a result. These observations however, may also be due to the presence of habitat forming species such as kelp or fucoids (Wernberg and Goldberg 2008; Schiel and Lilley 2011). These species can provide areas of shelter from daily environmental stressors, and exert strong influences on diversity and evenness estimates of associated community structures (Korpinen et al. 2010). This outlines the importance of determining species composition for ecological interpretation (González-Megías et al. 2011).

Taken together, macroalgae and faunal species diversity patterns showed different ecological responses throughout this study, which could be due to their individual response to the system's intrinsic or extrinsic properties. Wernberg and Goldberg (2008) describe that such ecological pattern inferences can be ambiguous or confounded, with observations a result of the natural variability of species' diversity. Other studies however, outline that combining different taxonomic groups within the same system for study can reveal key wide-scale observations (Okuda et al. 2010). This includes examining the direction of large scale environmental and biological processes across various levels (Díaz et al. 2011). Therefore the use of both taxonomic groups across the intertidal regions is important within this under-sampled environment, particularly for future natural or anthropogenic impact assessments, in the long term.

Species community composition

Macroalgae composition followed the observed diversity patterns, with variation occurring across different spatial and temporal scales within this marine tidal race environment. Interestingly, macroalgae composition within Longis Bay showed greater temporal similarity throughout the study, compared to the Control Bay. Previous ecological studies link macroalgae spatio-temporal compositional patterns to the interaction of several key abiotic and biotic factors, including substrate heterogeneity and wave exposure (Steneck and Dethier 1994; Okuda et al. 2010; Díaz et al. 2011). For example, I recorded increased proportions of *A. nodosum* and fewer crustose Rhodophyta species within Longis Bay, which are often associated with reduced wave exposure environments (Menge et al. 2005; Okuda et al. 2010; Burrows 2012). These two bays could therefore be influenced by such factors, differently. Alternatively, the two bays could be influenced by similar factors, with compositional differences due to the life history characteristics of the present macroalgae assemblage (see Appendix 3 for additional species composition SIMPER results). I found that Longis Bay comprised of larger proportions of ephemeral species, including *Ulva intestinalis*. This assemblage combination is a potential indicator of a natural or anthropogenic disturbed environment type (Steneck and Dethier 1994; Chapman 2007; Wells et al. 2007). This begins to outline the potentially different ecological processes occurring within these bays, which warrants future in-depth study.

Intertidal faunal composition within this system predominately comprised of molluscan species, which altered across the measured spatial scales. The recorded faunal taxonomic types and compositional differences observed may be a combination of the strong hydrodynamics and the macroalgae assemblage structure present. Korpinen et al (2010) showed that molluscan composition decreased with increasing exposure, which corresponded to the size of algae individuals. The study determined that molluscs had a reduced ability to stay attached to thalli of algae in exposed sites, due to macroalgae individuals exhibiting smaller sizes. Other studies have also shown the combined importance of exposure level with macroalgae composition type influencing faunal species' movement, grazing, reproduction and recruitment activities (Alfaro and Carpenter 1999; Guerry et al. 2009; Cacabelos et al. 2010). This further suggests the differing underlying environmental and biotic properties between the two bays, and across all shoreline heights within this type of environment. This may also account for the strong faunal compositional variation observed across the temporal annual and seasonal scales, due to these factors known to alter over time (Menge et al. 2005).

In-depth ecological studies are however, required to further determine these biotic relationships between macroalgae and faunal assemblages, and their interactions with the variable hydrodynamic conditions found within marine tidal race environments. This may enhance basic tidal velocity descriptions which are used currently for ecological studies, particularly in terms of temporal variation (Denny 2006).

Species colonisation

The annual processes of colonisation within the lower shoreline level of this system varied. My results also showed larger growth, diversity and differing species composition within Longis Bay. This suggests that the availability of settling spores or propagules for colonisation is much greater within this bay, compared to the Control Bay (Benedetti-Cecchi and Cinelli 1993). Species which attributed to this increased diversity within Longis Bay largely exhibited ephemeral life history tendencies, which are again linked to increased disturbance i.e. *Dilsea carnosa* (Wells et al. 2007). These species are also acknowledged as early colonists, with their presence commonly listed in previous succession research (Sousa 1979; Dean and Connell 1987a; Dean and Connell 1987b; Kim 1997; Patrício et al. 2006).

The pathways of succession have been researched extensively, with studies outlining a range of determining factors that can contribute to spatial variation (Chapman 2002). This is further related to the present disturbance regime, intrinsic vulnerability of resident communities and life history traits of the colonising species involved (Sousa 1984; Benedetti-Cecchi and Cinelli 1993). Macroalgae succession is largely dependent upon the physical attributes of the surrounding matrix; localised environmental influences and the present grazing assemblage (including biological attributes of algae for chemical or mechanical defence), with many studies highlighting the importance of temporal variation (Chapman 2007; Aguilera and Navarrete 2007; Cacabelos et al. 2010). In terms of temporal patterns, both bays separately showed comparative patterns of diversity and composition between the annual replacements of the experimental tiles. This suggests that when new or cleared patches occur at this point in time, the annual colonisation processes within these bays are similar, but differ spatially overall. This outlines the possibility of differing small scale processes influencing these two bays, and therefore the direction of species succession (Chapman 2007).

The source of available faunal colonists was largely represented by adult molluscs (personal observation), with similar estimates of diversity and composition observed between both

bays. A number of studies describe these particular adult species to emigrate from the surrounding area, and can show temporal variation during succession processes (Benedetti-Cecchi and Cinelli 1993; Chapman 2007). The Control Bay showed similar faunal compositions between the annual scales, unlike Longis Bay, where the composition changed each year. These observed subtle compositional differences may be a further reflection of the differing underlying intrinsic and extrinsic processes within these two bays overall.

Caveats, conclusions and future applications

The combined compositional and colonisation sampling strategies used in this study were useful as they outlined different aspects of ecological variation within this under-sampled environment. These combined methods could therefore be used for future studies within this environment type or other research objectives within the intertidal system i.e. monitoring anthropogenic activities or climate change impacts.

The study showed annual temporal diversity and compositional responses, which I related to the ecological and environmental processes, that are commonly known to occur within intertidal shorelines (Menge et al. 2005). The observed temporal variances however, may also be the result of in-balance between the sampling replication units (i.e. 2011) and influence the interpretation of the results given (GLM macroalgae and faunal diversity interaction variables between Year: Season and Bay: Year). Therefore careful consideration must be taken to some degree, with future studies to include more balanced designs where possible. In terms of general experimental design, my study also only included two bays within the intertidal system of the Alderney Race tidal race under review. Thus, incorporating additional bay sites would increase our ecological knowledge of these environments and sampling replication for univariate and multivariate analysis validation at the spatial scale. This could include adjacent shorelines to Alderney on the French coastline or other Channel Islands within the geographical area. In addition, this study (and results) could be integrated with other wider-scaled surveys or monitoring programmes of the intertidal environment. This could include: the MarClim UK and Ireland rocky shore surveys; the MarLIN Shore Thing surveys; the PISCO regional coastal study baseline data collection; the MARBEE rocky-shore ecosystem functioning project or the European Commission's VECTORS project. These projects investigate the ecological patterns of intertidal species related to ecosystem functioning, climate change, ocean acidification and human impact factors, at a broader scale. For example, the VECTORS project aims to investigate species distributional change (including invasive species) and productivity across changing environmental gradients (VECTORS

2013). A wider-scaled, comparative analysis and integration with these projects could first increase this study's sampling replication efforts, whilst further determine ecological patterns within this environment type. This may then help enhance current knowledge of intertidal ecological patterns across differing environmental gradients, biological processes and geographical locations (UK, Channel Islands and Europe), at a broader level.

Determining the causal intrinsic and extrinsic factors of the variances observed is difficult, first in terms of the wide range of possible factors to address, and second that some of these factors are highly interactive (Okuda et al. 2010). Therefore, including a more in-depth review of the present community structure may reveal such factors or other underlying stressors i.e. natural and anthropogenic influences (Martins et al. 2010). This could include identifying species life history trait information (functional form or life stage) and examining ecological communities at a wider scale, in terms of coarser taxonomic levels (such as family or sub-phylum levels) or the biotope/ habitat level (Steneck and Dethier 1994).

The colonisation experiment provided complementary ecological information. This study however, only provided a small 'snap shot' of the intertidal colonisation processes within these systems, in terms of temporal scales. The patterns of succession observed within both bays may well have changed if the experimental tiles had been left longer. Also, the timing of the deployment of tiles could have contributed to the observations on the type or life history stage of the present colonising species (Underwood and Chapman 2006; Chapman 2007). Colonists can show varying abundance and distributional patterns at the seasonal scale, related to factors such as recruitment (Turner and Todd 1993). Future studies could therefore consider deploying tiles for longer time lengths and/or implementing them at staggered time periods; to compare colonisation at the seasonal scale. The potential for the loss or breakage of tiles is however high (personal observation) due to the strength of the wintering storms exhibited in these systems. Therefore, I recommend the use of alternative methods (such as substrate clearance) or the use of different materials i.e. cement colonisation blocks (Antoniadou et al. 2011). Clearance methods could also allow for supplementary manipulation experiments within these environments, which are fundamental to marine ecology research (Underwood 2000).

Overall, I recommend further research including manipulation experiments, specifically within the mid and lower shoreline heights of these systems. These shorelines exhibited

considerable macroalgae and faunal ecological variation differences within a small distance, which may primarily be due to the graduated spatial and temporal levels of hydrodynamic intensities or other intrinsic and extrinsic factors within this system. These studies should incorporate additional physical-biotic comparative research methods, to examine the complex relationship and mechanism between the present hydrodynamics forces and ecological communities, in more depth. Such research could also improve the tidal velocity or wave exposure basic descriptive categorisations which are currently used for ecological study, whilst further enhancing ecological understanding of this under-sampled environment type. For example, future studies could examine inshore tidal velocity rates or wave exposure (coastline aspect, wave fetch and shoreline height profiles) with faunal species assemblage interactions, across different temporal scales. Alternatively, studies could help further test ecological theories, such as the intermediate disturbance hypothesis, through the assessment of the macroalgae assemblage composition and colonisation between the two shoreline heights over other temporal scales, using supplementary manipulation experimental methods (Dean and Connell 1978a; Tilman 1990). In addition, I also observed the presence of filter-feeding sponge and barnacle species during both the composition and colonisation studies, within the lower shorelines. Although these species occurred in low cover proportions (< 0.2 % total cover proportion recorded during composition study) within the studied bays, their inclusion into future physical-biotic studies may reveal additional important patterns within this type of extreme hydrodynamic environment.

Overall, this study revealed the variable degree of ecological spatial and temporal heterogeneity that can occur within the intertidal shorelines of marine tidal race environments. My results, combined with the outlined caveats and suggestions, now present a variety of future research possibilities for study within this highly intense hydrodynamic environment. These could further advance current ecological theories or give the potential to further define the physical and ecological components that occur within these extreme environments. In addition, studies could use these methods to examine the impacts of new anthropogenic activities such as renewable energy developments, which these environment types may face in the future. As such, this enhances the importance of the intertidal system as a valuable marine environment for ecological study in the long term.

Chapter 4. Short term temporal responses of pollack, *Pollachius pollachius* to a deployed marine tidal energy device.

4.1. Abstract

At present, few studies exist that consider temporal species' interactions and their relationship to key environmental variables alongside the added influence of anthropogenic activities. Here, I combine biological and environmental survey techniques to help to relate ecological responses to new offshore marine renewable energy technologies. I investigated the temporal response of Pollack, *Pollachius pollachius,* to the presence of an operational offshore tidal energy device and their overall relationship to velocity flow rates.

Trials were conducted by integrating video footage with ADCP survey techniques, across 15 day temporal periods, during the summer months of 2009 and 2010. Five random photographic stills were taken from the video footage at hourly intervals throughout each trial day to estimate mean hour fish abundance. Mean abundance was then compared between the hour and day temporal scales and their relationship with velocity rate for both years.

I observed that *P. pollachius* aggregated in shoals round the deployed device, with larger abundance observed in 2009 than 2010. Abundance was significantly associated to velocity rate for both trial years. Increased abundance was related to a reduction in velocity rate for both years, with shoals potentially using the device for temporary protection or feeding strategies. Responses to tidal velocity also differed between years, with 2009 abundances ranging from 0 – 1.2 m/s and 2010 abundances between 0.5 – 1.7 m/s. Overall the study outlined a useful integrated approach to investigate species responses with new anthropogenic activities and their relationships with key environmental site variables.

4.2. Introduction

In-depth knowledge relating to ecological responses to offshore marine tidal energy technologies is still lacking, as a result of gaps in ecological baseline information and, the tidal energy industry still at an early research and design phase (Gill 2005; Neill et al. 2009; Shields et al. 2009). The application to determine ecological response often involves combining environmental and biological study techniques, particularly to understand species' biological and behavioural trends to new anthropogenic activities within the marine environment (Hermant et al. 2010; Bosman et al. 2011). An advance in survey technology and experimental sampling designs has provided a variety of approaches to monitor marine species' ecological responses, including video photography and acoustic sampling techniques (Ehrhold et al. 2006; Ierodiaconou et al. 2011).

The scope for future tidal energy development schemes is increasing throughout the United Kingdom (UK); primarily due to the government's future renewable energy contribution lead target aims (Devine-Wright 2011). By 2020, the UK government aims for at least 30 % of its electricity to be generated from renewable energy sources, with 10 % potentially contributed by tidal energy development schemes alone (DECC 2009). At present, tidal energy research is centred round the development and deployment of single device machines, which are tested within offshore tidal energy development sites.

Tidal energy development sites are located within environments which exhibit strong hydrodynamic conditions, such as increased water movements (Gill 2005; Neill et al. 2009; Shields et al. 2011). Enhanced water movements are not only a fundamental environmental variable for tidal energy extraction, but are also known to strongly influence marine species' natural behaviour, presence and distribution (Hiscock 1983; Shields et al. 2011). Therefore, the inclusion of this key environmental variable to determine the ecological response to tidal energy development activities may be important (Shields et al. 2011). Water movement is commonly measured using velocity rate units (metres per second) using technology, such as acoustic doppler current profilers (ADCP) and have been applied to a number of biological, geophysical and oceanographical studies (Wewetzer et al. 1999; Reed et al. 2004; Kostachuk et al. 2005).

Current methods for the assessment of marine ecology within these extreme hydrodynamic conditions are generally limited to the use of video or photography camera systems for visual species observations and habitat classifications (Albert et al. 2003; He 2003; Lorance and

Trenkle 2006). Video techniques can measure valuable biological responses at both the temporal and spatial scale, and are not limited in terms of sampling time or weather restrictions (Monk et al. 2011). This includes species foraging or feeding behaviours, swimming activities, breeding observations and discrete habitat/species interactions (Lauth et al. 2004; Lorance and Trenkle 2006; Reubens et al. 2011). The use of video has been applied in comparable offshore environments, with recent studies evaluating the ecological interactions from other renewable energy structures including wind and wave devices (Wilhelmsson et al. 2006; Langhamer et al. 2010; Reubens et al. 2011; Shields et al. 2011). Such studies have examined direct biological population responses to renewable devices but few have investigated responses such as fish aggregation behaviour over temporal scales or environmental factors such as tidal velocity rate (Macpherson 1998; Tessier et al. 2005; Wilhelmsson et al. 2006; Stobart et al. 2007).

Knowledge of the spatial structure or aggregation behaviour in fish populations has become extremely important, in terms of commercial fisheries management strategies and environmental impact scenarios (Albert et al. 2003). In particular, a number of commercial and by-catch species are known to aggregate round natural and anthropogenic structures including boulders, kelp forests, coastal defence structures and offshore oil rigs (Pizzolon et al. 2008; Dempster et al. 2010). In this study, I focus on the species *Pollachius pollachius* (common name: pollack), which is also known to form shoals round a variety of natural and anthropogenic structures (Rangeley and Kramer 1995), following a preliminary study (2009). This species is a gadoid predator, common throughout the British Isles and found within a number of habitats which range from the rocky inshore regions to offshore sand regions (Sarno et al. 1994; Henderson and Bird 2010).

By combining video photography techniques with tidal velocity measurements, fish interactions in response to deployed tidal devices over temporal scales and their relationship with key variables may be identified. Such integrated methods could aid renewable energy environmental impact assessments and further ecological knowledge on species behavioural patterns (Shields et al. 2009; Langhamer et al. 2010). The overall aim for this study was to examine abundance responses of *P. pollachius* to a deployed tidal energy test device by (i) assessing *P. pollachius* abundance responses over temporal hour, day and year scales and (ii) examining *P. pollachius* abundance relationship to tidal velocity rates. This was undertaken through experimental trials combining underwater video observations with ADCP velocity rate measurements. For this chapter, I test the hypothesis that the deployed tidal energy test

device has the potential to act as an artificial reef structure and thus an aggregation site for species such as *P. pollachius*. I also test that their ecological response to the deployed device is irrespective of the measured temporal scales and environmental factors, such as tidal velocity rate.

4.3. Methods

Study area

Trials were conducted within the European Marine Energy Centre (EMEC) offshore tidal test site, located off the coast of the Isle of Eday, Orkney Isles. The test site is situated within the Fall of Warness marine tidal race environment which is approximately 2 km wide and 3.5 km long (Norris and Droniou 2007). The stream has an average depth of 20 – 35 metres, with tidal flow movements from both the North-West and South-East within a daily tidal cycle (Osalusi et al. 2009). The OpenHydro Ltd tidal device platform is installed at the most northern part of the test site (59°09.448`N, 02°49.561`W; Figure 4.1). This device is a sub-tidal open turbine generator, consisting of a 6 metre diameter turbine mounted on a twin mono-piled platform. The platform is placed into the seabed, producing a footprint approximately 10 m² (OpenHydro Ltd 2008). It should be noted that this site had limited fisheries activities in the past (Personal communication: Orkney Fisheries Association 2009). In addition no other commercial fisheries vessels or activities were known to occur within this site throughout the timescales of the study (Personal communication: EMEC 2009). The trial surveys were conducted between the summer months of June and July in 2009 and 2010. I chose these months based on the ease of access to the device platform and the potential for sampling problems in other periods of the year, due to weather conditions. These were selected in collaboration with EMEC and Open Hydro Ltd personnel. Collecting the data from the platform is known to be highly difficult during the spring and winter months due to the unfavourable weather conditions and also the potential risk of losing the data during collection. Surveys were conducted across 15 day trial periods, which I also selected based on access to the device platform. The 2009 trial began at the end of June, with the 2010 trial at the start of June. The 2010 trial lost nine days of video footage and ADCP recordings after day seven. This was due to a weak electrical cable link between the recording devices and the device platform. The fault was identified during the survey and fixed straight away, with the survey extending for a further nine days to account for the missing data.

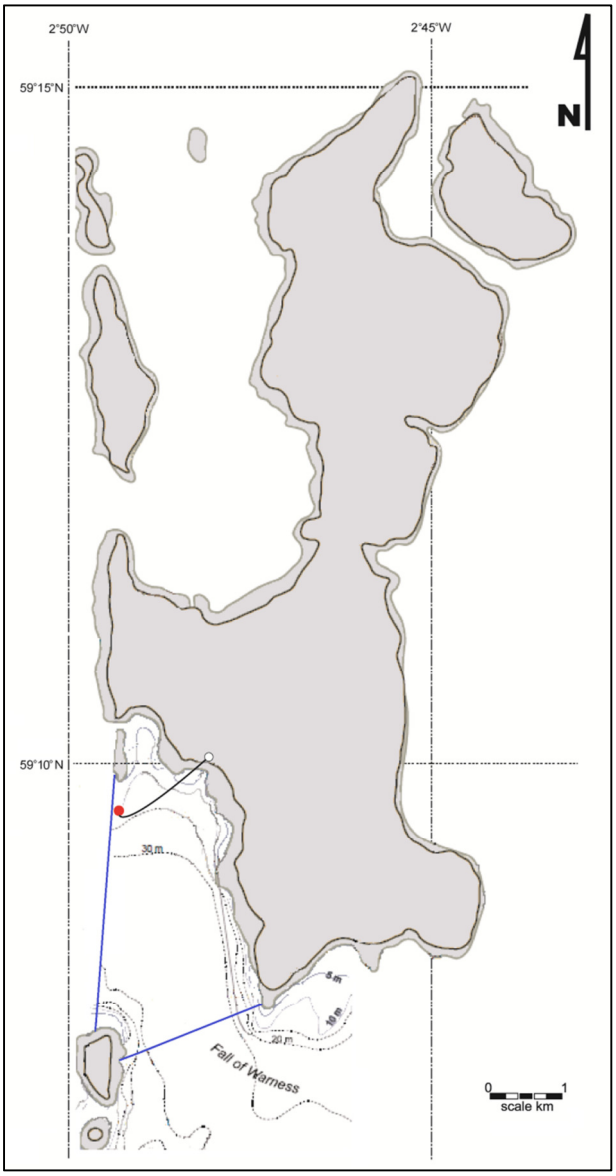


Figure 4.1. Location of the EMEC tidal energy test site and OpenHydro Ltd deployed tidal device, situated off the South-West coast of the Isle of Eday, Orkney Isles. The extent of the EMEC tidal energy test site facility is within the blue lines connected to the Isle of Eday and surrounding islets. The red circle and joined black line represents the position of the deployed OpenHydro Ltd test platform device and sub-sea cable route (adapted from Aurora 2005).

Video Fish Observation Sampling Method

The underwater footage was recorded using a video Triplex 8 Channel DVR, linked to a Submertec Video Camera System mounted to the outside of the OpenHydro Ltd platform

device (Figure 4.2). The video camera system was mounted approximately 2 metres from the face of the turbine allowing continuous recording of the entire 6 metre turbine area. This system recorded continuously throughout the surveys and was switched off after the last survey day for both years. I then collected the video footage manually after the full trial period each year and transferred it to a compatible video computer Programmable Logic Controller (PLC) software system.

I extracted five randomly chosen photographic still frames from the first two minutes of footage from each hour of the survey data, recording the hour, day and year of each image. This technique follows other timed animal behavioural methodologies that use photographic techniques to assess species abundances (Shucksmith et al. 2006). This gives an estimate of relative abundance, whilst reducing the chance of counting individual organisms twice, which can occur during video assessments (Becker et al. 2010; Becker et al. 2011; Birt et al. 2012). Random photograph selection within the two minute time period was determined by the second timeframe status, using a random number generation application in the R statistical software package (R Development Core Team 2010). Due to the extremely long summer daylight hours at this latitude, I was able to undertake the analysis throughout the 24 hour clock cycle. During both survey years however, footage was excluded where light levels or weather conditions affected the quality of the photograph still, and where the field of view was obscured by the presence of marine algae debris.

I counted the maximum abundance of *P. pollachius* individuals within each sampled frame. I then calculated the geometric mean across the five individual frames sampled within each hour, which included zero counts. Fish were identified visually from the photograph frames based on body shape, lateral line and mouth part descriptions where possible and only individuals identified unequivocally to the species level were included within the analysis (Hayward and Ryland 1995).

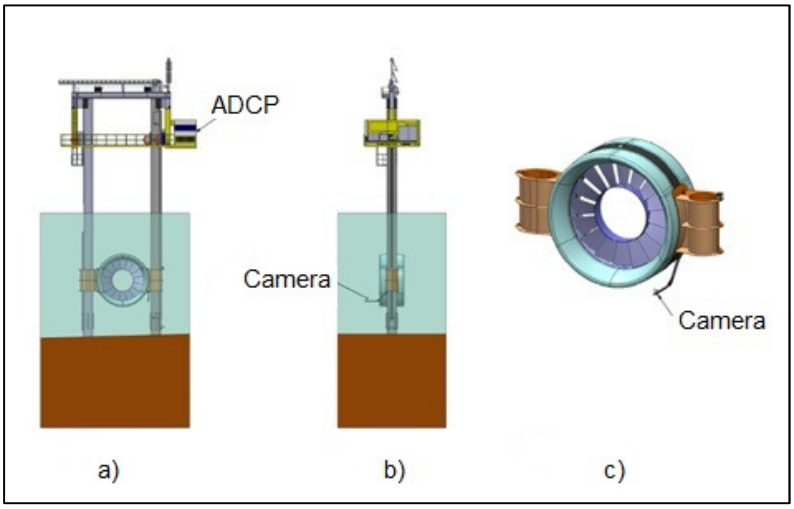


Figure 4.2. Schematic diagram of the front view a), side view b), and turbine face view c), of the video camera system and ADCP system setup attached to the deployed OpenHydro Ltd tidal energy test device platform within the EMEC tidal energy test development site, Isle of Eday.

ADCP Velocity Rate Measurements

A Nortek Aquadopp two beam ADCP was deployed at the platform in order to measure horizontal tidal current flow around the turbine and through the Fall of Warness over a 24 hour cycle. The two beams were deployed either side of the platform to measure flow on flood (North) and ebb (South) tides (Figure 4.2). Data was streamed live to the PLC which is used for control and monitoring of the turbine and the tidal velocity was sampled every 10 seconds and recorded on a PC. I collected the data at the end of each annual trial and then downloaded into a Microsoft Excel format for analysis, with velocity measured in metres per second (m/s). Hourly harmonic mean tidal velocity rates were then calculated across 10 second samples for both the North and South ADCP velocity measurements for each annual trial. In order to account for the change in direction in the tidal flow, the higher of the two hourly means from the North and South ADCP measurements were used in analyses.

Data Analysis

Analyses were conducted using the R statistical programming software (R Development Core Team 2010). I used generalised linear models (GLM) and also generalised additive models

(GAM) to investigate fish abundance response with different temporal scales and the tidal velocity variable for both annual trials. These methods are useful for examining ecological count data and defining relationship patterns between variables (Crawley 2007).

For each trial year I used a GLM to investigate the mean hour fish abundance with the temporal scales; hour and day. The categorical explanatory variables for this model were the time of day (assigned to the 24 hour clock) and the day trial number, with the fish abundance count as the response variable. These analyses were then further extended to examine fish abundance in relation to velocity rate using a GAM, separately for each year. The continuous explanatory variable was tidal velocity, with the response variable as mean hour fish abundance. This analysis was implemented using the R library function 'mgcv' with all variables smoothed as a function with all interactions considered within the overall GAM models ('mgcv' Cran package, from Wood 2006). These GAMs used the lowess non-parametric smoother to view the overall environmental-biotic relationship (Cleveland 1981; Crawley 2007). GLMs were also used to compare mean hour fish abundance between the 2009 and 2010 trials, with year as the categorical explanatory variable and fish abundance as the response variable. The GLM and GAM models used the Poisson distribution of errors (family = Poisson, link function = log) and assessed in terms of homogeneity throughout visual inspection of the Q-Q plots (Crawley 2007). The quasi-poisson error structure was also used to deal with over-dispersion, where the residual deviance is greater than the residual degrees of freedom in the fitted model. This error structure frees the model from specifying a specific distribution but maximum likelihood and likelihood ratio tests cannot be used (Crawley 2007). The significance of the explanatory variables within each GLM or GAM was deduced by comparing models with and without the chosen variable term using analysis of deviance with the F test. Variables were deemed significant based on the increase in deviance from their resulting removal from the model following model deletion test methods ($p > 0.05$) (Crawley 2007).

4.4. Results

Video Observations

I recorded the presence of *P. pollachius* within both of the 2009 and 2010 survey trials, and no other marine species. Across the total number of hours recorded (n = 261), *P. pollachius* contributed 13% to the total proportion of fish presence during 2009 and 8% to total fish

presence in the 2010 trial. The 2009 trial estimated larger levels of fish abundance, with a total sum of 664 individuals recorded. This trial also recorded the largest range of abundance within each trial hour, which ranged from 0 – 46 (mean count per hour = 44). The 2010 trial recorded a total sum of 121 individuals, with abundance ranging from 0 – 11 per hour (mean count per hour = 7). Overall, fish abundance during the 2009 trial fluctuated across the time of day (24 hour clock), and trial day, with no significant temporal relationships identified (Table 4.1; Figure 4.3). The 2010 GLM results showed a significant relationship with the hour and day scales. During this trial year I observed no counts between the hours of 20:00 – 05.00, and between the trial days 6 – 9 respectively.

Table 4.1. Generalised linear model (GLM) F ratio results for the temporal scales; hour, day, and year of fish abundance for the 2009 and 2010 trials, Isle of Eday. All models based on family = quasi-poisson distribution, link = log function, n = 261 (per year).

GLM	Explanatory model variables	*F*-ratio	p-value
2009	Hour	1.266	0.219
	Day	1.17	0.29
2010	Hour	2.81	< 0.001
	Day	2.22	0.007
Year		25.60	< 0.001

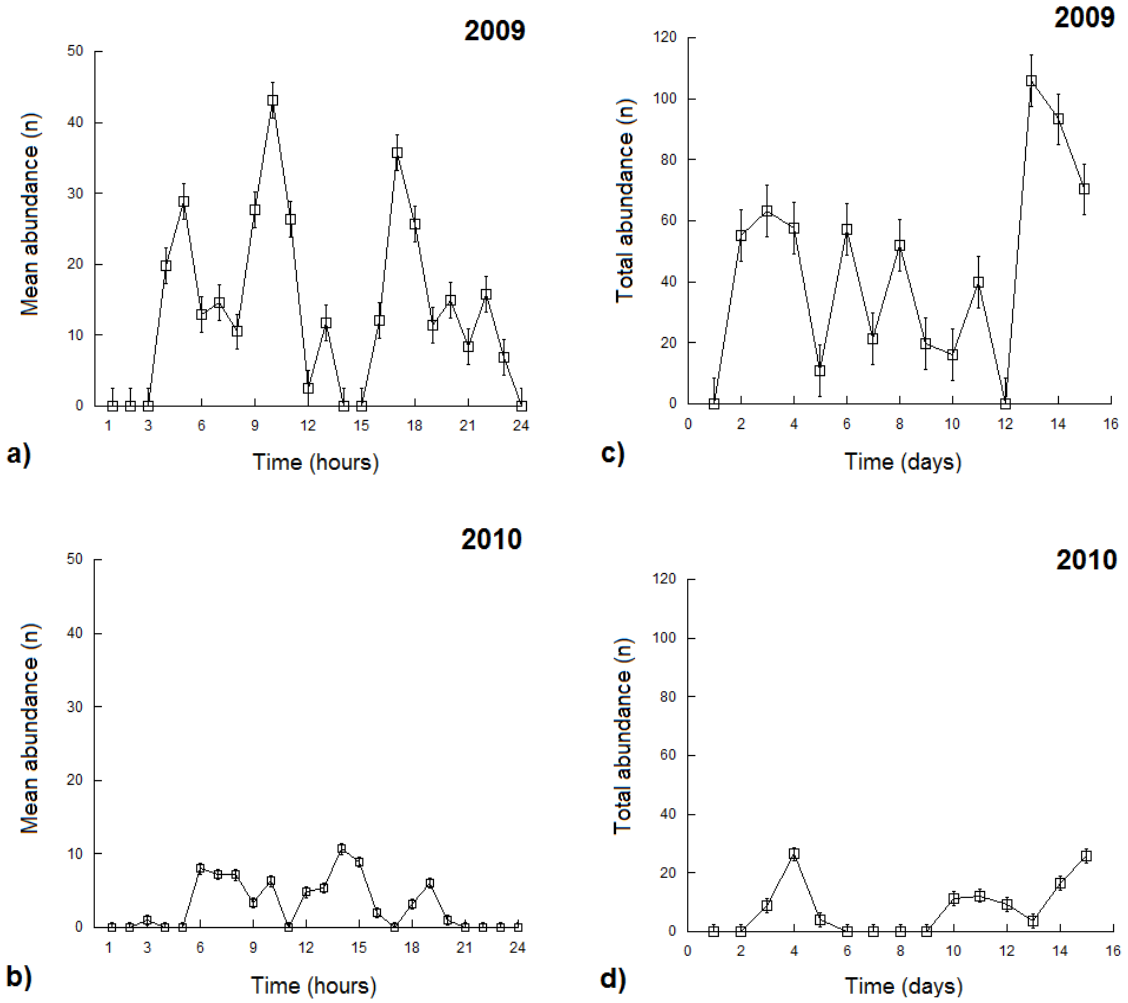


Figure 4.3. Mean fish abundance per hour (a – b) and total fish abundance per trial day (c – d) recorded during the 2009 and 2010 video survey trials (± 1 SEM), Isle of Eday. Mean abundance per hour is the combined abundance counts within each hour across the trial day, with hour based on the 24 hour clock. Total abundance per trial day is the combined total of the mean hour fish abundance counts for every 24 hour survey trial day.

ADCP Velocity Rate Surveys

The ADCP tidal velocity rate trial surveys identified a larger velocity rate range in 2010 than 2009 overall. In the 2009 trial, the velocity rates ranged from 0.260 – 2.780 m/s, with the highest velocity rate observed in day 9 and the lowest rate in day 14 (Figure 4.4). The largest range of velocity rates occurred in days 8 and 9, and the lowest range in days 2 and 13. The 2010 trial showed velocity rate to range from 0.180 – 3.05 m/s, with the highest velocity rate observed in day 10 and the lowest in day 15. The largest velocity rate range occurred during

days 9 and 11, with the lowest range in days 2 and 7. This pattern generally mirrored the two separate time periods across the overall survey.

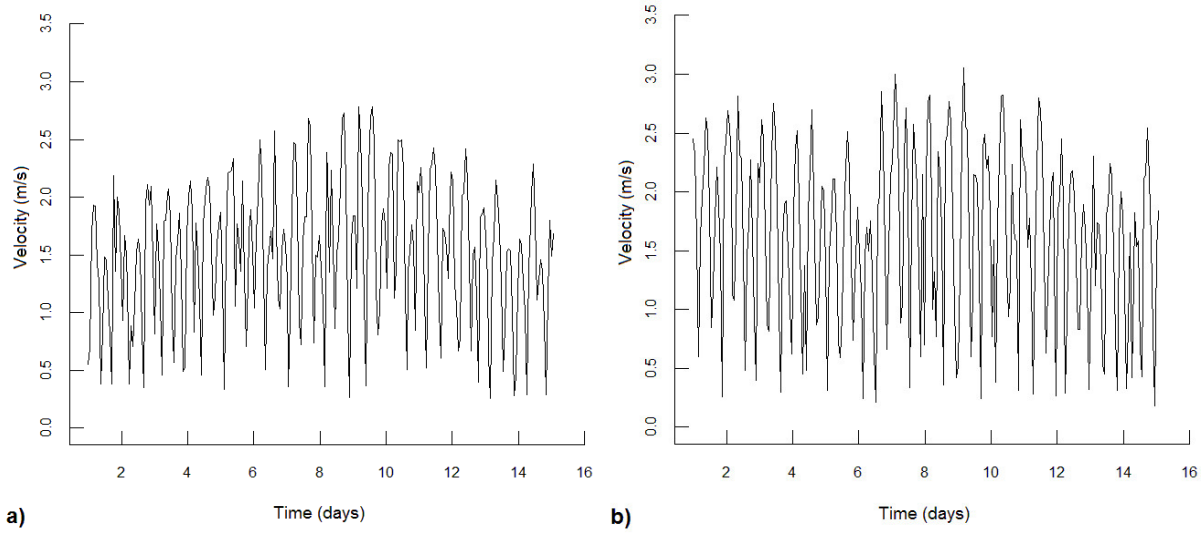


Figure 4.4. ADCP velocity rates across the total survey day trials for a) 2009 and b) 2010, Isle of Eday. Velocity is measured as metres per seconds (m/s), with measurements recorded in 10 second intervals throughout the 15 day survey trials.

Fish Abundance Response to Tidal Velocity

The GAM models for both trial years outline a significant relationship with fish abundance and velocity rates ($GAM^{2009VELOCITY}$: F = 40.78, $p < 0.001$, n = 261; $GAM^{2010VELOCITY}$: F = 4.45, $p = 0.01$, n = 261). I observed that in both year trials, fish abundance declined as velocity rate increased. The pattern of this relationship however, was different for both years (Figure 4.5). During the 2009 trial, I observed fish abundance to rapidly decline as velocity rate increased. Fish abundance was observed to occur largely between velocity rates of 0 to 1.0 m/s, with few observations of fish presence after 1.3 m/s. During the 2010 trial I observed fish presence to predominately occur between the larger velocity rates of 0.5 - 1.7 m/s and then decline above 1.8 m/s.

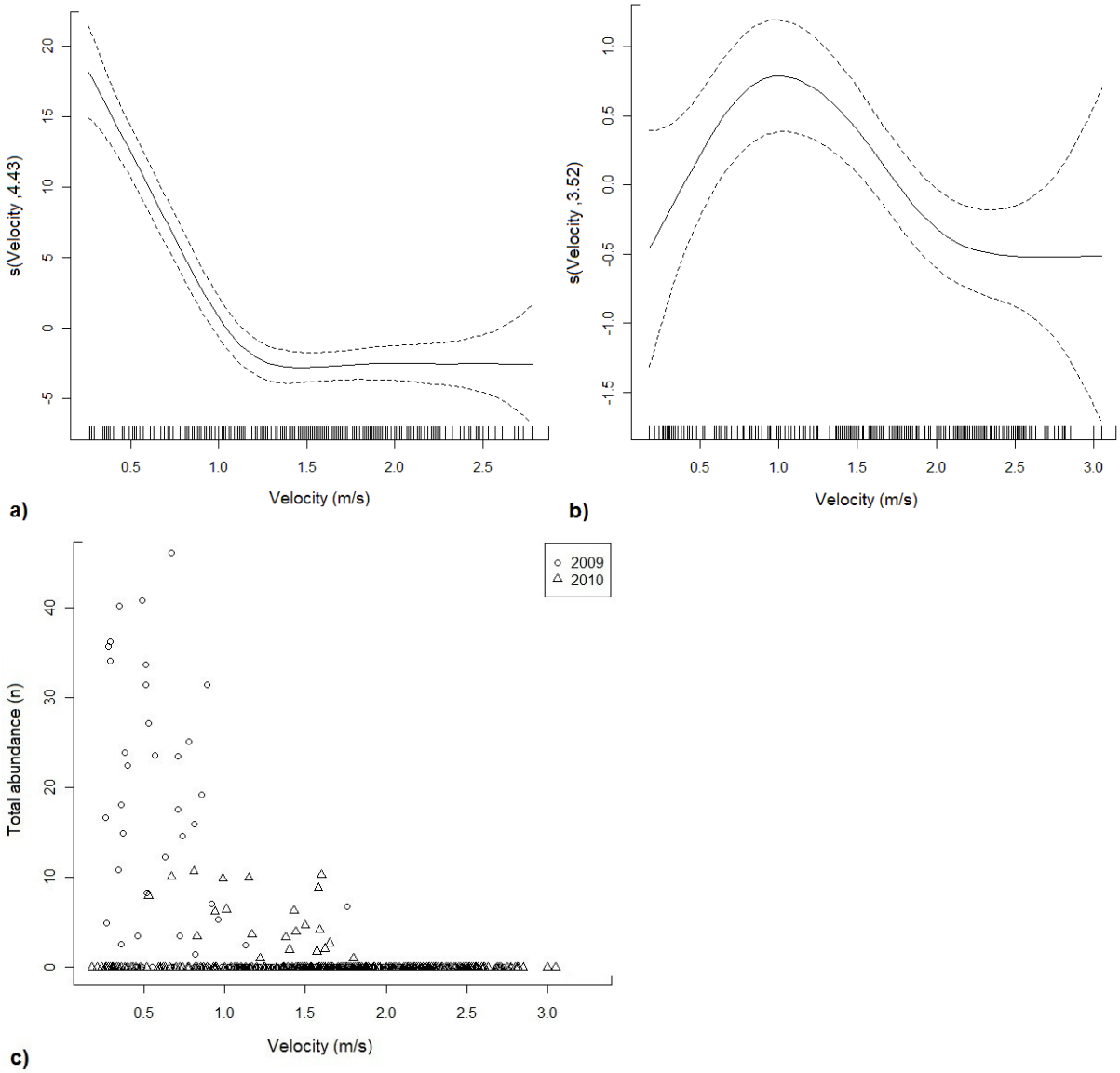


Figure 4.5. Generalised additive models (GAM) of the relationship between *P. pollachius* abundance and the measured velocity rate for a) 2009 and b) 2010, and the total individual abundance counts recorded across velocity rate for both years c), Isle of Eday. For a) and b), dashed lines represent the 95% confidence intervals. For c) circles represent the 2009 survey trial, with triangles representing the 2010 survey trial abundance counts.

4.5. Discussion

Video Observations

The 2009 and 2010 video survey trials recorded the presence of *P. pollachius* surrounding the deployed tidal energy device, and no other recorded marine species. My results outlined that for both years, fish predominately occurred in groups, with few observations of solitary individuals. Grouping or shoaling activities are a common behavioural trait in fish species, providing individuals with the potential for increased feeding, spawning and predator avoidance (Cohen et al. 1990; Froese and Pauly 2007; Rowley 2008; Seppälä et al. 2008). The predatory feeding strategy of *P. pollachius* is to remain stationary in the water column and use natural (reefs and marine algae) or anthropogenic structures (breakwaters) to strike out at passing prey (Froese and Pauly 2007; Pizzolon et al. 2008). Therefore I suggest that the deployed tidal device could offer new aggregation sites for species such as *P. pollachius*, particularly in terms of providing potential feeding or refuge sites. This follows other studies which outline that offshore renewable energy structures act as artificial reefs, attracting a variety of species for such activities (Wilhelmsson et al. 2006; Langhamer et al. 2010; Reubens et al. 2010). I also observed that fish presence was relatively small, in comparison to the total recorded video footage within both years. This species is known to remain within the vicinity of local waters, which suggests that the observed temporary aggregations were potentially from the same group, repeatedly returning to the device during the trials (Cohen et al. 1990; Sarno et al. 1994). It was not possible however, to identify if this was the case, but does suggest the future need for new electronic fish tagging and tracking experiments.

Video observations showed considerable variation in fish abundance across the temporal scales, with observed interaction during the 2010 trial. Fish populations are known to fluctuate naturally, with daily behavioural cycles also influenced by biological and environmental cues including prey or predator movements, water depth, water direction, salinity, or anthropogenic influences (Sarno et al. 1994; Cargnelli et al. 1999; Seppälä et al. 2008; Trancart et al. 2011). Comparisons between the two trials also outlined significant differences in fish observations, with increased abundances seen in the 2009 trial overall. These variations are often linked to annual factors of temperature change or spawning events (Cargnelli et al. 1999; Selleslagh and Amara 2008). Alternatively, the variation between the two years could also be due to the population structure, in terms of aggregation fish age/ size, as group structure is often related to fish size i.e. juvenile nursery aggregations or grouped adult spawning events (Le Fur and Simon 2009). Including measures of these factors in

future trials may outline the causal nature in the patterns of variation observed both within and between these measured temporal scales. In addition, fish abundance response patterns can also occur at monthly or seasonal temporal scales, which were beyond the scope of these preliminary trials. Therefore an overall extension of the trial's time period length and annual surveying is recommended, to outline any potential temporal trends to the deployed device. These trials could also be linked to other available fisheries information relating to the population status of this species within the regional area. Current regional fisheries information on *P. pollachius* is limited, as this species is regarded as a non-commercial fisheries species, often caught as a 'bycatch' species (ICES, 2012a; ICES, 2012b). Alternative sources of information could be found from independent scientific studies, marine biological recording programmes (such as the National Biodiversity Network) or amateur angling data within the region. This would provide in-depth species population dynamic information, which could be applied to assess temporal factor inferences and wide-scale ecological interactions with new anthropogenic activities, such as those outlined in this study. In addition, this could include implementing comparable aggregation studies using these methods with other offshore structures including wave renewable energy devices, oil and gas platforms, offshore coastal defence structures or sea fish farms. For example, Dempster et al (2010), show how fish farms attract aggregations of 'wild' fish species (Cod, Saithe and Haddock) over different spatial and temporal scales. In addition, Wehkamp and Fischer (2013), suggest that new coastal defence structures act as artificial reefs and attract fish species from the surrounding area. Future wider-scaled studies or meta-analyses of such investigations could further reveal comparative fish aggregation patterns, and in turn aid wide-scale assessments of anthropogenic activities within the marine environment.

Tidal Velocity Fish Response

The observed increased velocity rates and temporal variation are comparable to past independent ADCP velocity rate surveys, with the local environment known for substantially strong and varied tidal flow conditions (Norris and Droniou 2007). This is due to the land masses of the Isle of Eday and Muckle Green Holm creating a natural narrow channel constricting the tidal flow, which leads to the observed increased velocity rates (Norris and Droniou 2007). In-depth surveys within these types of environments are exceedingly uncommon, due to the complexity of the physical seabed characteristics and the degree of

variation of these strong hydrodynamic conditions (such as turbulence), restricting survey effort (UKHO personal communication 2009; Bailly du Bois et al. 2012). As such, velocity rates or tidal range information (including Chart Datum) only provides an initial, basic view-point of the system or generalised predictions, and may therefore mis-represent the true tidal characteristics and hydrodynamic properties (Bailly du Bois et al. 2012). In addition, there is limited freely available information to determine the tidal characteristics of the Fall of Warness site (nearest port locations for tidal characteristic information: Isles of Egilsay or Rapness, Orkney) such as tidal amplitude and phase, although commercial data from tidal energy developers may exist (UKHO personal communication 2009; 2013). Therefore, these results and the use of ADCP methods also further aid key information on the nature of these environments, in terms of velocity profile descriptions. The GAM models portrayed comparative relationships between fish abundance and velocity rates surrounding the deployed tidal device for both annual trials. Significantly, few abundance counts of *P. pollachius* were observed at high velocity flow rates, with increased abundance counts related to low tidal velocity rates. A number of studies suggest that certain fish species use a 'selective tidal stream transport' mechanism, whereby selecting tidal flows to actively migrate across the water column (both horizontally and vertically streams within the water column) (Wroblewski et al. 2000; Trancart et al. 2011). Therefore, the large velocity rates observed may drive *P. pollachius* aggregations away from the deployed device to other local regions and structures, to areas of better protection or feeding grounds (Selleslagh and Amara 2008). As tidal velocity declines, shoals may then be more inclined to move away from these areas and aggregate round the tidal stream device.

The ADCP surveys for both years identified subtle tidal velocity curves, which could be related to periodic (monthly) 'spring' and 'neap' tidal pattern conditions. These tidal cycles could strongly influence fish abundance within the area, with resulting spring tidal conditions reducing the number of fish observations and neap tidal conditions increasing them. For example, Wroblewski et al (2000), describe how strong tidal flow events (such as spring tidal conditions) significantly alter fish migration patterns, direction and general movements. Trancart et al (2011), also show that periodic tidal current patterns (such as ebb and flood tidal cycles) can influence the swimming activities of fish species, such as the Thinlip mullet (*Liza ramada*). This species showed increased swimming activities during flood tidal cycles, which were related to the species' feeding activities. Further comparisons to such periodic tidal pattern currents (such as: spring vs. neap and ebb vs. flood cycles) and other tidal

characteristics could therefore accurately define the changing patterns of fish abundance responses to the tidal energy device and the overall tidal characteristics of the site.

During the 2010 trial, fish observations were seen to occur at larger velocity rates than the populations observed in the 2009 trial. Shoals are known to aggregate over different spatial scales, influenced by the complexity of physical structures, habitat patchiness and the natural behaviour of the species or population involved (Le Fur and Simon 2009). *P. pollachius* also portray opportunistic trait tendencies and are found in a variety of pelagic, benthopelagic and estuarine environments and often regarded as marine migrant opportunists (Henderson and Bird 2010; Li et al. 2010). Overall fish abundance responses surrounding the tidal device could be described as temporary opportunistic aggregations, responding to local abiotic factors such as tidal velocity conditions, which change annually.

I also identified a proportion of unexplained variation between abundance and tidal velocity rates for both annual trials ($^{\delta}$ (from analysis of deviance) 2009 = 35 %, $^{\delta}$ 2010 = 86 %), with the 2010 trial outlining considerable unexplained variation. This may be due to the study's overall sampling effort/ design. The video camera system's field of view could also not view the area around the bottom of monopole structures or the seabed itself. Fish aggregations could therefore have occurred surrounding or below the device legs, behind the camera system itself or further outside the camera system's field of view. These factors should therefore be taken into consideration and assessed in more detail in terms of increasing the video/ camera system sampling efficiency or integrating other survey techniques into the experimental trials i.e. inclusion of other environmental variables (sea surface temperature), electronic fish tagging studies and additional camera units on the device platform legs (Selleslagh and Amara 2008). It should also be noted that the 2010 survey trial lost a number of video survey days which could therefore attribute to the high unexplained variation value, which is further shown by the wider spread of the confidence intervals and the level of interaction in the GLM and GAM results.

The unexplained variances may also be the result of other direct or indirect biotic, abiotic and anthropogenic factors, not included in the study. For example, the study by Trancart et al. (2011), also describe that thinlip mullet swimming activities, whilst strongly influenced by tidal flow, are also affected by other environmental cues such as wave action, light intensity and day period, and their interaction (i.e. the day or hour period: day vs. night). The study highlighted that this particular fish species showed active movements during the light (day)

phases, and reduced activity in the dark phases (night). Overall, the study revealed that this species uses a diurnal flood tide transport to actively migrate and move. The unexplained variation and level of interaction shown by the GLM and GAM model results may therefore be highlighting the combined interactions between velocity rate, other environmental cues (wave action or meteorological patterns) and tidal cycles or characteristics (neap/spring or ebb/flood patterns and their interaction), which occur at different temporal scales, upon my fish response observations. Therefore, future studies should include and asses the interactions between these other environmental parameters and tidal characteristics across different temporal scales to fully determine the behavioural responses of fish to tidal energy devices.

Combined Survey Trial Approach: Caveats and Future Considerations

Both the ADCP and video/ still photography sampling procedures are ideal for such extreme environments and renewable energy device assessments due to the devices offshore location, cost effectiveness, reduced sampling bias and increased sampling effort (Wewetzer et al. 1999; Reed et al. 2004; Krag et al. 2009; Monk et al. 2011). Video and still photography assessments are useful when examining marine species across many temporal or spatial scales, but are ultimately a tool that requires further application, with other techniques including acoustic sampling for sound method design (Diesing et al. 2009). Therefore in the context of the survey aims, this combined approach was useful outlining preliminary information of renewable energy device interactions with a specific marine species. The survey method could be used for further species behavioural investigations, or other renewable energy device interaction surveys and marine management applications i.e. fisheries and conservation activities.

More method testing between the sampling procedures, temporal scales and statistical analyses is however now required. This includes expanding the survey sampling effort in terms of increasing the number of generated photograph stills, length of survey trials and additional camera systems attached to the other side, further down the device legs or, placed at graduated distances away from the device. For example, fish are known to show vertical migrations to the seafloor during dark (night) phases, following changes in light intensity, temperature and salinity, for protection purposes (Trancart et al. 2011). Extra camera systems placed further down the device platform legs may therefore reveal such diurnal behavioural patterns or other unrecorded behavioural responses (including other marine species) to the

device. Camera systems placed at different distances from the device could also reveal the temporal presence, strength, size and pattern of the aggregations (Dempster et al. 2010). In addition, the use of electronic fish tagging and tracking methods would also be extremely useful to determine the overall distributional patterns of fish within the area (Wroblewski et al. 2000). This would increase knowledge of fish presence, behaviour and local movements surrounding the device and across the local environment, which is currently unknown. Other environmental factors and analysis should also be included in future surveys such as light levels, salinity, tidal cycle period state and characteristics, sea temperature, wave climate and meteorological information (Osalusi et al. 2009; Hermant et al. 2010). This also includes monitoring other anthropogenic activities within the overall tidal test site during the survey trials such as vessel activities, pile-driving, device deployments and other research activities. Such natural and anthropogenic variables may heavily influence fish presence and behaviour and also reduce the observed variance found within my preliminary statistical models.

Overall, these preliminary results show fish behavioural responses to one deployed tidal energy device within this particular marine environment and geographical location. Other ecological responses (fish presence and behaviour) by marine species (including other marine species) may occur with other tidal energy device developments (including device life cycles and array size), within marine environment types (marine or estuarine) and geographical locations (temperate or tropical, proximity to coastlines) (Dempster et al. 2010). In addition this may also include environments which undergo different scenarios, such as commercial fishing practices. A number of studies suggest that governmental regulatory legislation will prohibit fishing activities within areas selected for tidal energy developments (Gill 2005; Inger et al. 2009; Dempster et al. 2010). As such, fish presence and biomass may increase within the vicinity of tidal energy device developments, particularly within locations which may have experienced fishing activities in the past (Dempster et al. 2010). Therefore, further studies should emphasise on including additional control sites and tidal energy device developments, across different environments, locations and scenarios. This will enhance survey replication efforts and further validate these preliminary results. It will fundamentally outline in-depth, comparative ecological responses to tidal energy developments at a wider scale. This is particularly important given the potential conservation benefits and large scale developments predicted of this type of marine renewable energy source in the future (Gill 2005).

Conclusions

The experimental trials outlined *P. pollachius* observed behaviour to aggregate temporally surrounding the deployed renewable energy tidal stream device during 2009 and 2010 trials. Fish aggregations fluctuated considerably across hour and day temporal scales, with temporal interactions found in 2010. Tidal velocity was identified to influence the presence of fish aggregations, with increasing tidal velocities seen to clearly reduce the number of observations. Fish aggregations were not observed above 1.3 m/s in 2009 trials and 1.8 m/s in 2010 respectively. Overall this combined experimental method identified preliminary responses of local species interactions with renewable energy devices in the marine tidal race environment. Additional method testing and assessment of the experimental trials is recommended to increase sampling efficiency for future applications to other devices or survey sites.

Chapter 5. Fine-scale benthic assemblage response with a deployed marine tidal energy device.

5.1. Abstract

The addition of man-made structures to the marine environment is known to increase the physical complexity of the seafloor, which can influence the biodiversity and assemblage structure of benthic communities. We lack however, specific knowledge of the ecological influences of deployed tidal energy device structures. In this chapter, I examine the effects on benthic species biodiversity, assemblage structure and habitat type around a single deployed test scaled device within the EMEC tidal energy test site, Orkney. I use a combination of commercial potting methods and benthic towed video camera techniques over three temporal sampling periods during 2009 and 2010.

My results showed increasing biodiversity estimates and changes in assemblage composition within the vicinity of the deployed device. These were attributed to the added presence of species, such as *Homarus gammarus*, potentially using this area for refuge or feeding opportunities. I found that these sites largely comprised of crustacean species, omnivore or predatory feeding regimes and tide-swept habitat types. I associated these ecological components with the extreme hydrodynamic conditions found within this environment. My results also suggest that these ecological components alter over the time, which I related to natural environmental and biological processes, such as storms, reproduction and recruitment events. As such, I conclude that the deployed device could act as an artificial reef for local benthic assemblages, but that further in-depth and long term studies are required. This would be beneficial, given the scale of future commercial tidal energy development projects (> 100 devices; > 1 km^2) within the marine environment as a whole.

5.2. Introduction

The regions of the marine benthic zone support a variety of species communities and habitats, which are influenced by a complex interplay of chemical, physical and biological factors (Dounas et al. 2007; Ierodiaconou et al. 2011). A number of marine species are known to actively select particular benthic habitat types, with preferences based on life history strategies. This includes recruitment, migration, refuge and feeding requirements, which can change throughout an individual's lifetime (Moksnes 2002; Pardo et al. 2007).

Past studies indicate that the physical complexity of the bottom/ seafloor topography creates a diversity of benthic habitats, which are exploited by a range of organisms (Ierodiaconou et al. 2011). The addition of new physical structures to the seafloor can therefore increase physical complexity and habitat diversity with such structures acting as 'artificial reefs'. Artificial reefs provide additional ecological niches, leading to new trophic interactions and increasing the overall biomass, biodiversity and increased opportunity for reproductive or recruitment events within the local marine environment (Langhamer 2010). These structures, however, can also alter both local seabed topography and the hydrodynamics of the surrounding waters and hence dramatically change benthic community structure and species distributions at a local scale (Page et al. 1999).

Deployments of offshore marine renewable tidal energy devices within the benthic zone have the potential to act as additional artificial reef structures and thus alter communities and habitats. This has been shown in recent ecological impact studies of wind turbine and wave energy deployments, with devices affecting localised species assemblages (Wilhelmsson and Malm 2008; Langhamer and Wilhelmsson 2009; Langhamer et al. 2009). Knowledge of how tidal energy devices affect the benthic community structures is patchy. This is due to the relative novelty of the tidal energy industry and also the difficulty of sampling within marine environments selected for development (Gill 2005; Shields et al. 2011).

Sites allocated for tidal energy developments are situated within marine environments with extreme tidal hydrodynamic conditions. This can limit the overall choice of ecological field sampling methods or constrain sampling effort to narrow time windows around slack water. A number of survey methods can be adapted to work within these extreme conditions, including commercial fisheries potting practices, bathymetry mapping and benthic video photography techniques (Ierodiaconou et al. 2011; van Rein et al. 2011). Such methods have been used for a variety of ecological studies in comparable offshore locations and provide

good, quantitative information on benthic species community composition (Dunnington et al. 2005). This includes population studies of mobile benthic species (crustacean abundance and diversity assessments) and distributional studies of non-mobile benthic species (macro-algae, hydroid or sponge distribution assessments) which are regarded as the main component of the benthic habitat structure (Martin et al. 2005; Post et al. 2011).

The aim of this chapter is to examine the benthic ecology within a tidal energy development site, including benthic species biodiversity, community composition and trophic functionality, using commercial fisheries potting practice methods. These can give good quantitative information and indications of environmental change within an environment and provide information on natural or anthropogenic variation within these mobile benthic assemblages (Martin et al. 2005; Wilhelmsson and Malm 2008).

I also include a benthic habitat assessment using benthic video photography techniques, as these can help complement in-depth ecological compositional studies (Pardo et al. 2007). For example, benthic habitat assessments are regarded as valuable ecological assessment tools for ecosystem management and marine spatial planning activities (Shumchenia and King 2010). The assessment of benthic habitats consists of examining both mobile and non-mobile species present and their distribution, within a selected survey area (Brown and Collier 2008). Benthic habitats are determined qualitatively, using descriptive classification schemes, such as the EUNIS benthic habitat classification scheme (Connor et al. 2004). These classifications also incorporate other biological, physical, and chemical information for defining habitat type, such as seabed bottom topography (water depth and substrate type), hydrodynamic properties (wave exposure and tidal velocity rates) and human impacts (marine litter and fishing practices) present (Shumchenia and King 2010; Brown et al. 2011). Therefore, the inclusion of benthic habitat assessment, combined with in-depth compositional studies may reveal other important characteristics of this marine environment type and anthropogenic impact under study.

Within the tidal energy development site, I focus primarily on exploring the ecological influence of an existing deployed single tidal energy test device. Understanding ecological interactions with single test devices can begin to quantify how commercial tidal energy device structures may interact with the marine environment (Shields et al. 2011). Such information is essential, particularly in light of the large scale tidal energy development schemes planned for UK waters in the future (Bahaj 2011). Both natural and anthropogenic

processes within the regions of the benthic zone are also known to operate and influence at more than one spatial and temporal scale, with these effects displayed across different assemblage levels (Pardo et al. 2007; Orlando-Bonaca et al. 2008). Therefore this study explores the benthic community structures across different scales such as temporal seasonal sampling periods in order to reveal variation in both the ecology of the benthic environment and anthropogenic impacts (Antoniadou et al. 2011). Here, I test the hypothesis that the deployed tidal energy test device alters the benthic ecology present within the device site, compared to the control site. I also test the hypothesis that this anthropogenic influence will occur regardless of the measured temporal scales.

5.3. Methods

Study area

The study area was located within the benthic regions of the Fall of Warness tidal race environment, adjacent to the Isle of Eday, Orkney Isles (Figure 5.1a). The Fall of Warness tidal race environment is approximately 2 km wide and 3.5 km long, with average depths from 20 – 35 metres (Norris and Droniou 2007). Tidal flow movements alternate between North-West and South-East directions within the daily tidal cycle, with peak mean flows ranging from 7 knots (3.5m/s) during spring tides and 2.85 knots (1.44 m/s) during neap tides (Osalusi et al. 2009). Sub-tidal substrates within this area are described as predominately exposed bedrock, boulders, cobbles and coarse gravels (Admiralty Nautical Charts 2007; Aquatera Ltd 2008). Within the study area, I selected two offshore benthic survey sites, regarded as the device and control sites (Figure 5.1b). The device site was located on the South-West coast of the Isle of Eday, within the EMEC tidal energy test site facility. This site enables manufacturers to assess device performance for future commercial production and manufacture, and provides placements for single test or commercial scaled devices. During this study, the OpenHydro Ltd tidal energy device test platform was the only permanent machine deployed within the EMEC site, however the rest of the site was in use by EMEC, OpenHydro Ltd and other tidal energy developers (personal observation). This device was located at the most northern part of the EMEC test site (Figure 5.1; 59°09.448`N, 02°49.561`W). The test device is an underwater generator consisting of a 6 metre diameter open turbine mounted on two pile leg structures placed into the seabed, with a footprint of approximately 10 m², deployed in 2006 (Figure 5.2; OpenHydro Ltd 2008).

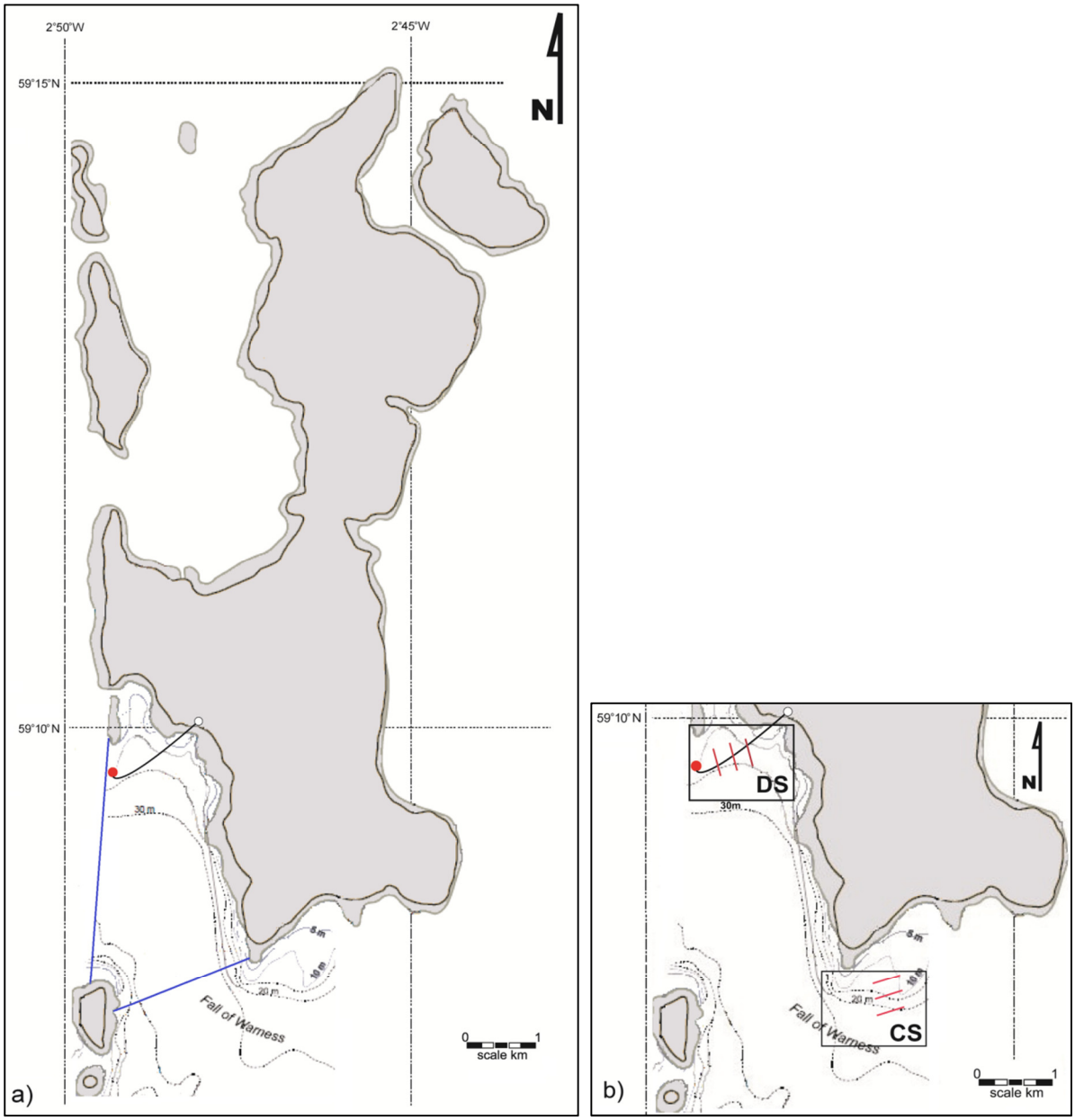


Figure 5.1. Location of the benthic assemblage study and habitat assessment, from 2009 – 2010, Isle of Eday. a) The Fall of Warness tidal race environment on the South-West coast of the Isle of Eday. The total extent of the EMEC tidal energy test site facility is shown within the blue lines. The red circle and joined black line represents the position of the deployed OpenHydro Ltd test device and sub-sea cable route. b) The two survey sites, the device site (DS) and the control site (CS), with the positions of the survey potting transect string lines represented by red lines. Each string line is 500 metres long and placed 200 metres away from each other, facing parallel to the Isle of Eday coastline. Maps are adapted from Aurora (2005).

The control site was located on the South-East coast of the Isle of Eday and chosen for ecological comparisons. I selected this site based on its similarity to the device site in terms of physical and ecological properties, such as seabed bathymetry, water depth and tidal velocity conditions. This site is not part of the EMEC tidal test site, but is still within the Fall of Warness tidal race environment. It should be noted that both sites had limited commercial potting fisheries activities in the past (Personal communication: Orkney Fisheries Association 2009; 2013). In addition no other commercial fisheries vessels or activities were observed within these sites throughout the timescales of the study.

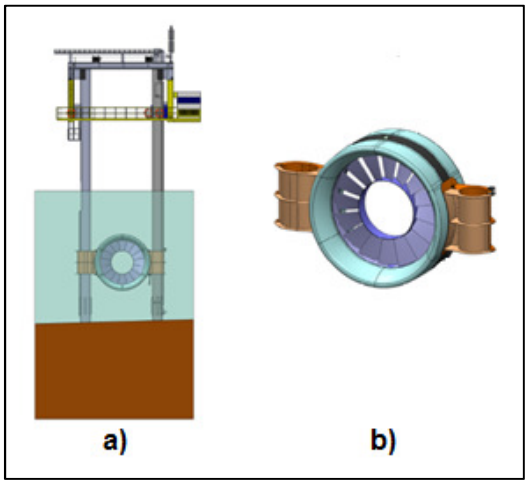


Figure 5.2. Schematic drawing of the OpenHydro Ltd sub-sea tidal energy device deployed at the EMEC test facility site, Isle of Eday. a) Frontal view of the twin piled test structure platform at sub-sea level (water depth approximately 30 metres) and b) the 6 metre diameter turbine face.

Mobile benthic species assemblage: commercial potting assessment

I assessed benthic species assemblage composition within the device and control survey sites using 500 metre commercial crustacean potting strings (regarded as string transect lines for this study), deployed from a commercial fishing vessel.

In the device site, sampling entailed deploying three 500 metre (0.27 nautical miles) string transect lines, at varying distances, horizontally adjacent to the deployed OpenHydro Ltd tidal energy device test platform and the coastline of Eday (Figure 5.1b DS labelled area).

String transect lines were placed approximately 200 (recorded as string 1, nearest to the device), 400 (string 2) and 600 (string 3, nearest to the coastline) metres away from each other on the Eastern side of the device. I then repeated this method in the control site (Figure 5.1b CS labelled area); with the string transect lines positioned at the same distances adjacent to the Isle of Eday coastline. The location and mean water depth of each string transect line was recorded using on-board vessel differential GPS (dGPS) and acoustic echo-sounder information (see Appendix 4 Table A.4.1 for string transect line GPS starting positions).

Each individual 500 metre string transect line comprised of 35 numbered (1-35) 'D' shaped commercial creel pots. Pots were attached approximately every 10 - 12 metres along each string transect line and buoyed with a surface marker at one end. All string transect lines were first deployed in both survey sites during the same voyage trip using an on-board winch system from the fishing vessel. These were then left to 'soak' for 36 hours, hauled (retrieved), ecologically assessed (on-board the vessel) and then re-deployed, repeatedly for a total of 4 sessions (every 36 hours). Each repetitive string transect line haul within both survey sites was conducted during the same voyage trips. Pots contained the same bait type (half fresh *Pollachius pollachius*), with all pots re-baited during every haul.

Ecological assessment entailed randomly selecting 10 pots (out of the 35 number pots) during each repetitive haul (4 hauls) within each string transect line (1; 2; 3) and survey site (device; control) and recording the species caught per selected pot. Random pot selection was undertaken using pre-generated random numbers, from random number generation techniques in R (numbers ranging from 1-35) (R Development Core Team 2010). I counted and identified all caught individuals within these selected pots to species level; using species' keys for species diversity and composition assessments (Table 5.1; Hayward and Ryland 1995; Hayward et al. 2001). For taxonomic and functional form composition assessments, I combined the species count data into corresponding phylum and feeding regime categories using species reference texts and biological databases, such as the online BIOTIC database (Table 5.1, MarLIN 2006; Appeltans et al. 2012).

All surveys were conducted during the slack period within neap tidal cycles in order to avoid the high levels of equipment movement, breakage or loss associated with extreme spring tidal flow in such sites.

I conducted this sampling method within both survey sites across three temporal seasonal sampling periods: October 2009, April 2010, and October 2010. A fourth survey scheduled for July 2010 was abandoned due to extreme weather and hydrodynamic conditions.

Benthic habitat: video assessment

I examined the benthic habitats of the study area in terms of physical substrate and ecological species cover composition along each string transect line, within both survey sites using towed camera video techniques (following Brown and Collier 2008; Sánchez et al. 2009 and see Appendix 2 Table A.2.1 and Figure A.2.1 for camera system setup). This assessment was only undertaken during the April 2010 sampling season, due to the difficulty of deploying camera equipment within the October survey seasons, as a result of the increased hydrodynamic regimes and weather conditions that can occur during these temporal periods. Prior to this survey, I identified broad substrate type and water depths for both survey sites using Chart Datum and bathymetry charts provided by EMEC (Admiralty Nautical Charts 2007).

The towed camera video system was deployed at the start of each 500 metre string transect line location, with the vessel controlled to drift at a speed of between 0.5 – 1.7 knots along each line. The camera was held approximately 1 metre above the seabed from the vessel, using the on-board commercial fishing winch system. This was fixed facing downwards to continuously video record the seafloor onto the video camera system's hard-drive and an external hard-drive, along with dGPS recorded every second using SeaTrack^{TM}. The camera was fitted with a 4.3 mm wide-angle lens, recording at an area of approximately 1.25m^2.

From the recorded video data of each string transect line within both survey sites; I extracted twenty photographic stills using Adobe Photoshop photographic software (Adobe 2012). These images were selected at random using randomised number generation techniques in R, based on the photograph's GPS decimal/ minute coordinates (ranging from 0 - 60 minute coordinate numbers) (Service and Golding 2001; Coggan et al. 2005; MESH 2008). I then visually identified physical substrate and ecological species cover type and their proportion within each still frame. The physical substrate type was classified using a generalised version of the Wentworth 1922 sediment scale, which included an additional organic material category (Table 5.1; Greene et al. 1999; Howell 2010). The cover species were identified

taxonomically to the phylum level using species reference keys (Hayward and Ryland 1995; Brown and Collier 2008). Hydroid and bryozoan species were difficult to separate using video data and were therefore grouped together as the category 'Other'. Physical and ecological proportions were assessed in terms of their total percentage cover (%) within each image and measured using the 'ImageJ' photographic software (ImageJ 2012).

Table 5.1. Ecological and physical categories used in the mobile benthic assemblage and benthic habitat assessments from 2009 – 2010, Isle of Eday. The benthic assemblage assessment includes the taxonomic and feeding regime type classifications used during the commercial potting survey. The habitat assessment includes the physical (substrate grain size) and ecological substrate (cover species) categories used during the video camera still analysis. The ecological category 'other' is based on a combination of bryozoan and hydrozoan species.

Assemblage Assessment		**Habitat Assessment**	
Taxonomic categories	**Feeding regime categories**	**Physical substrate categories**	**Ecological categories**
Chordata	Herbivore	Bare bedrock	Macroalgae
Crustacea	Omnivore	Boulder (grain size 0.25 - 3 m)	Porifera
Echinodermata	Predator	Cobble (grain size 64 – 256 mm)	Other
Mollusca	Suspension feeder	Pebble (grain size 4 – 64 mm)	
		Gravel (grain size 2 – 4 mm)	
		Sand (grain size < 2 mm)	
		Organic material	

Data analysis

Data were analysed using R statistical software (R Development Core Team 2010), with all exploratory multivariate routines applied using the Primer v6 software package (Clarke and Gorley 2006).

To assess species diversity and community assemblage composition, I took the average of individual counts of species for each haul separately (4 hauls), for each string transect line number (strings 1; 2; 3), within each survey site (device; control) and temporal seasonal sampling period (October 2009; April 2010; October 2010).

I calculated the species richness (*S*), Shannon-Wiener diversity (H',$\log_e$) and Pielou's evenness (J'), for each string transect line haul within the survey sites and temporal seasonal sampling periods, using the Primer v6 DIVERSE routine (n = 72). I then used generalised linear models (GLMs) to compare these diversity measures as a function of string transect line number, survey site, temporal seasonal sampling period and their second and third order interactions. The string transect line number, survey site and temporal seasonal sampling period were treated as categorical variables. I used quasi-poisson distribution errors for species richness data (S) to account for overdispersion in the count data and Gaussian distribution errors for Shannon-Wiener and Pielou's evenness indices. Non-significant interaction terms and main effects ($p > 0.05$) were removed from the overall models (using ANOVA with appropriate F or Chi tests), to produce minimum adequate models (Crawley 2007). Models were visually assessed for unequal variance and non-normality of residuals and, where this occurred, the response variable was log transformed.

To determine the composition of species, taxonomic and feeding regime types across the spatial and temporal scales, I first fourth root transformed the species, taxonomic and functional form compositional datasets prior to multivariate analyses. This transformation was used to down weight the contributions of the common and rare species, within the groups and feeding regimes samples (Clarke and Warwick 2001). The choice of this transformation was selected to view the ecological datasets in more depth and based on the visual inspection of each species and group (taxonomic and feeding regime) datasets by draftsman plots generated in Primer v6.

I used analysis of similarities (ANOSIM in Primer v6) to identify significant compositional differences between the string transect lines, survey sites and temporal seasonal sampling periods for the species, taxonomic and feeding regime assemblages (Langhamer et al. 2009). This routine is based on Bray-Curtis similarity resemblance matrix distances between samples (Legendre and Legendre 1998; Clarke and Warwick 2001) and compares the ranks of distance values within and between samples grouped by a factor. The analysis produces an R value, which ranges from -1 (high differences within groups) to 1 (high differences between groups) and uses random permutations of the grouping factor to assess significance. Separate one-way ANOSIM routines were first run 'within' each survey site with the string transect line number selected as a factor on the species and group datasets. I then ran two-way analyses with the survey site and the temporal seasonal sampling period selected as exploratory factors. Where significant differences between factor levels were found, I used

the SIMPER procedure to explore which species and groups accounted for these differences. This routine calculates a percentage contribution of each species within a pair of groups to the Bray-Curtis distances between those groups (Clarke and Gorley 2006). The contribution similarity percentage for each species sample was based on ≥ 90 % of the total percentage contribution, to omit any rare species or small contributions. To determine the composition 'within' each string transect line separately for both survey sites, I first ran these routines as one-way SIMPER analyses. To determine the overall composition for each survey site and temporal seasonal sampling period, I ran the routines as two-way SIMPER analyses. For the taxonomic and feeding regime analyses, I selected 100% of the total percentage contribution, due to the small number of variable categories (Clarke and Gorley 2006).

To examine the overall structure of the recorded benthic habitats, I first determined the mean percentage cover (%) of each physical substrate and ecological categories and standard deviation across the 20 still images for each string transect line within each survey site (MESH 2008; van Rein et al. 2011). I then used separate one-way ANOSIM analyses to explore the compositional difference of the combined substrate components' (physical and ecological categories) percentage cover between the three string transect lines and the two survey sites. A hierarchical cluster analysis was also used, to compare the overall composition of these combined substrate categories across the string transect line numbers and survey sites. This procedure was generated using the Primer v6 CLUSTER routine to produce a dendogram, with group average linkage selected. The ANOSIM and CLUSTER routines were based on square root transformations, with the combined substrate dataset normalised prior to analysis. Euclidean distance was used to generate the distance matrix, since this is considered appropriate for mixed environmental datasets (Legendre and Legendre 1998; Clarke and Warwick 2001).

I then selected generalised qualitative benthic habitat descriptions for each string transect line in both survey sites, using a combination of the video camera survey photography stills, raw video footage, bathymetry maps and water depth information (Connor et al. 2004; van Rein et al. 2011). Habitat classifications followed the descriptive EUNIS hierarchical habitat structure classification schemes, which are based on the combination of physical and ecological sea-floor characteristics (Connor et al. 2004; Davies et al. 2004; EEA 2004; Brown and Collier 2008).

5.4. Results

A total number of 10,889 individuals and 14 species were recorded during the 2009 – 2010 benthic assemblage study, which comprised of 6 Crustacea, 3 Echinodermata, 4 Chordata and 1 Mollusca species (see Appendix 1 Table A.1 for species list). The benthic habitat assessment outlined a variety of physical substrates and ecological categories, which aided the classification of three benthic habitat types.

Mobile benthic species assemblage: commercial potting assessment

Species diversity measures

Species richness (*S*) was comparable between the string transect lines within both survey sites separately, with significantly higher diversity estimates found within the device site (mean *S*: device = 7.22; control = 3.33; Table 5.2). Across the three temporal seasonal sampling periods, I observed increased species richness during the April 2010 sampling period. I also found that within the device site, species richness increased across the measured sampling periods, whilst in the control site, diversity declined during October 2010 (Figure 5.3).

Similar patterns were also observed with the Shannon-Wiener diversity (H') measure, with comparable estimates between the string transect lines within both survey sites, and significantly larger diversity in the device survey site (mean H': device = 1.80; control = 1.22). Differences between the three temporal seasonal sampling periods were identified, with April 2010 and October 2010 sampling periods showing significantly increased diversity. I also observed different seasonal Shannon-Wiener diversity patterns between the two survey sites. In the device survey site, the April 2010 and October 2010 showed largely similar diversity, whilst in the control site, diversity lowered in October 2010 season.

Overall, the Pielou's evenness (J') diversity measure showed high, comparable values between the string transect lines, survey sites (mean J': device = 0.920; control = 0.934) and temporal seasonal sampling periods.

Table 5.2. Generalised linear minimal adequate model (MAM) results for benthic species diversity measure responses between the device and control survey sites and temporal seasonal sampling periods, from 2009 – 2010, Isle of Eday. Species diversity measures include species richness (*S*), Shannon-Wiener (H', $\log_e$) and Pielou's evenness (J') diversity between the two survey sites and temporal seasonal sampling periods (Oct 09; April 10; Oct 10). F represents the overall F model value; P represents the model P value; DF represents the degrees of freedom; n.s. represents no significant variables (n = 72). The Shannon-Wiener diversity model is based on a log transformation.

Model explanatory variable	F value	P	DF
Species richness (*S*)			
Intercept			
Site	24.71	<0.001	70
Season	7.56	0.005	69
Shannon-Wiener diversity (H')			
Intercept			
Site	23.68	<0.001	70
Season	8.23	0.004	69
Pielou's evenness (J')	n.s		

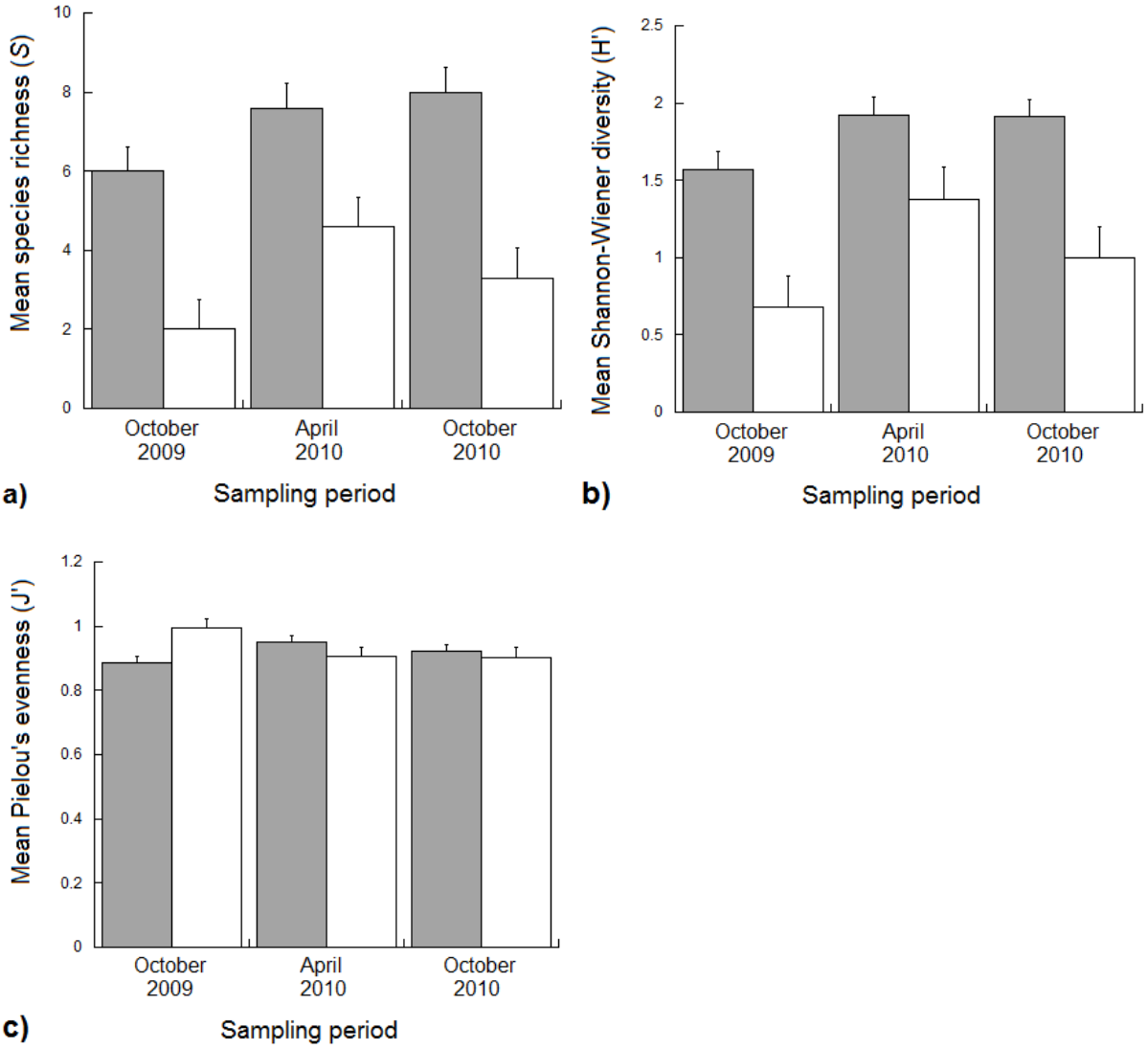


Figure 5.3. Mean benthic species diversity measures of the device and control survey sites across the three temporal seasonal sampling periods from 2009 - 2010, Isle of Eday. Grey bars represent the device survey site, with white bars representing the control survey site (± 1 SEM). Diversity measures are a) species richness (*S*), b) Shannon-Wiener diversity (H', $\log_e$) and c) Pielou's evenness (J').

Benthic assemblage composition

The composition of the benthic species and their corresponding taxonomic and feeding regime types were highly similar between the string transect lines within the survey sites, indicating that these sampling areas contain consistent biological communities (see Appendix 4 Table A.4.2 for additional string transect line SIMPER results). The ANOSIM results however, outlined significant differences in species, taxonomic and feeding regime composition between the survey sites and across the temporal seasonal sampling periods (Table 5.3).

The species composition SIMPER results showed that the ANOSIM differences between the two survey sites were due to the device site comprising of additional species, such as *Homarus gammarus* and *Hyas coarctatus* (Table 5.4). The SIMPER results also outlined that the seasonal differences were attributed to the fluctuating proportions of species including *Asterias rubens* and *Buccinum antiqua.* Overall, the SIMPER analysis outlined the crustaceans, *Necora puber* and *Cancer pagurus* to dominate the species assemblage structure.

The taxonomic SIMPER results revealed that the ANOSIM differences were a result of the device site comprising of the Chordata phylum, compared to the control site (Table 5.4). Both sites also comprised of large proportions of the Crustacea phylum. The seasonal differences were a result of the Echinodermata, Mollusca and Chordata phylum's varying proportions in April and October 2010, shown by the SIMPER analysis overall.

The feeding regime composition ANOSIM differences were associated with the larger proportions of predators in the device site, identified from the SIMPER results (Table 5.4). These SIMPER results also outlined the presence of herbivores during October 2009 season, with large proportions of predators during April 2010 season, accounting for the ANOSIM seasonal differences.

Table 5.3. Results of the ANOSIM tests for benthic species composition a), taxonomic composition b), and feeding regime composition c), between the string lines, survey sites and temporal seasonal sampling periods recorded during the benthic assemblage survey, from 2009 - 2010, Isle of Eday. String line comparisons are based on one-way ANOSIM analyses between the string transect lines within each survey site separately. Survey site (device; control) and temporal seasonal sampling periods (October 2009; April 2010; October 2010) comparisons are based on a two-way ANOSIM analysis, with season pairwise R value comparisons all significant ($p < 0.05$).

	Global values	
	R	***P***
a) Species		
String line: Device Site	-0.43	0.99
String line: Control Site	-0.3	1.0
Site	0.84	0.003
Season	0.683	0.001
b) Taxon		
String line: Device Site	-0.13	0.7
String line: Control Site	-0.3	0.96
Site	0.76	0.002
Season	0.679	0.001
c) Feeding regime		
String line: Device Site	-0.24	0.90
String line: Control Site	-0.11	0.59
Site	0.741	0.002
Season	0.691	0.001

Table 5.4. SIMPER percentage contribution (%) results of benthic species composition a), taxonomic composition b), and feeding regime composition c), within each survey site and season recorded during the benthic assemblage survey, from 2009 – 2010, Isle of Eday. Values are derived from two-way SIMPER analyses for each composition, site and temporal seasonal sampling periods, with average within-group similarities shown. Species composition results are based on 90% cut off, with 100% for taxon and feeding regime composition results. Values ≥ 20% shown in bold.

	Site		**Season**		
	Device	**Control**	**October 2009**	**April 2010**	**October 2010**
a) Species					
Crustacea					
Cancer pagurus	**27.99**	**42.93**	**41.44**	**32.49**	**32.62**
Homarus gammarus	8.89		4.46	4.87	3.66
Hyas coarctatus	6.41			10.11	
Necora puber	**32.98**	**51.52**	**46.73**	**34.81**	**45.50**
Pagurus bernhardus					6.08
Echinodermata					
Asterias rubens	8.95				4.24
Mollusca					
Buccinum undatum	5.49			8.99	
Average Similarity	83.39	88	89.82	82.55	84.71
b) Taxon					
Chordata	6.64		2.57		7.35
Crustacea	**67.78**	**89.57**	**88.80**	**65.35**	**81.65**
Echinodermata	17.86	6.49	8.63	19.48	8.34
Mollusca	7.81	3.94		15.17	2.66
Average Similarity	89.03	91.46	94.08	89.01	87.64
c) Feeding regime					
Omnivore	**73.44**	**89.44**	**88.77**	**71.01**	**84.93**
Predator	**24.82**	10.56	8.61	**28.99**	15.07
Herbivore	1.74		2.62		
Average Similarity	94.34	94.03	94.09	97.06	91.41

Benthic habitat: video assessment

The physical substrate components recorded within the two survey sites predominately consisted of gravels and pebbles (Table 5.5). In terms of ecological components, I identified large stands of the kelp *Laminaria hyperborea*, other macroalgae species, sponges and a mixture of unidentified turf species covering bedrock. Overall, the ANOSIM results outlined similar substrate compositions between the string transect lines ($R = 0.22$, $p = 0.4$) and survey sites ($R = 0.185$, $p = 0.2$). The cluster analysis confirms that the strings are not strongly clustered by site based on benthic habitat (Figure 5.4), suggesting that these two sites are generally comparable.

I visually identified three benthic habitat types within both survey sites, classified from the EUNIS hierarchical habitat classification system, out of a potential 28 circalittoral rock and sublittoral sediment habitat types (from codes A.4. to A.5., not including sub-category EUNIS level 6 scales (EEA 2004); Table 5.6). The habitats were classified as: kelp with cushion fauna and/or foliose red seaweeds; circalittoral mixed sediments and, *Tubularia indivisa* on tide-swept circalittoral rock. These habitats were found within the same string line numbers across both survey sites. The kelp habitat was found within strings 2 and 3 (nearest to the coastline), and the tide-swept habitat identified in strings 1 and 2 (furthest from the coastline). The mixed sediment habitat was identified across all strings at both survey sites.

Table 5.5. Mean physical and ecological substrate compositions identified within the device and control survey sites, from the towed video camera photographic stills recorded during April 2010, Isle of Eday. Physical and ecological epifaunal substrate proportions are given as mean percentage cover (%) and standard deviations (SD) across the string transect line numbers 1 – 3, for both survey sites.

	Location			
	Device		**Control**	
Substrate categories	**Mean (%)**	**SD**	**Mean (%)**	**SD**
Physical component				
Bare bedrock	2.5	2.29	11.5	8.67
Cobble (grain size 64 – 256 mm)	9.83	14.5	9.5	7.36
Pebble (grain size 4 – 64 mm)	14.33	8.25	11.84	11.47
Gravel (grain size 2 – 4 mm)	**42.83**	19.52	**22.5**	9.73
Ecological component				
Macroalgae	**22**	8.78	**21.33**	3.21
Porifera	4.53	7.33	12	4.09
Other	4	1.60	11.33	4.16

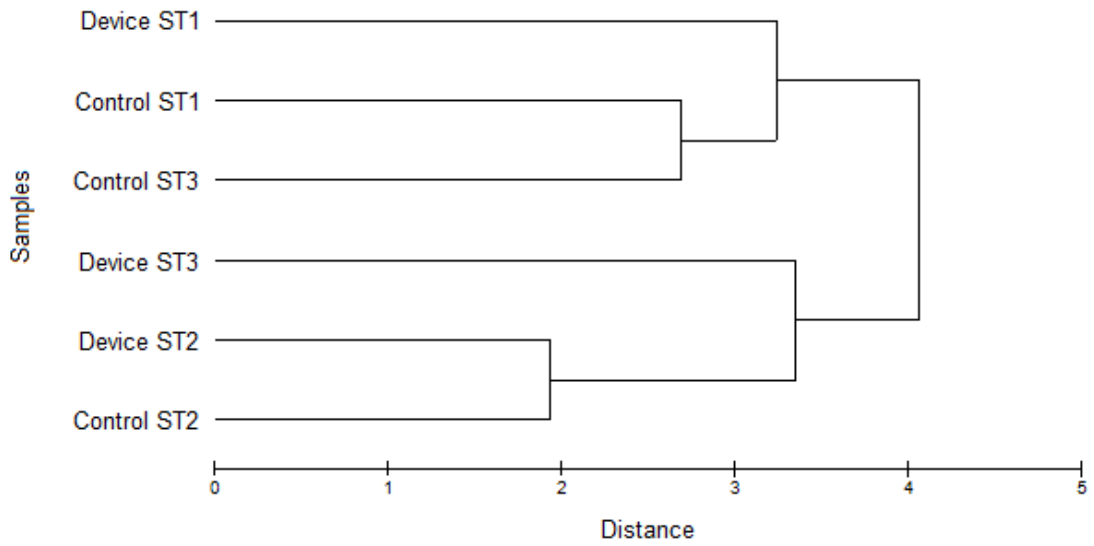


Figure 5.4. Hierarchical cluster analysis dendogram of the combined physical substrate and ecological components for each string transect line number, across the device and control survey sites, April 2010, Isle of Eday. Lettering corresponds to the survey area, ST1 – 3 represents the string number. Cluster analysis is based on the Euclidean Distance similarity matrix with group average linking.

Table 5.6. EUNIS marine habitat classification types identified within the device and control survey sites, from the towed video camera photographic stills recorded during April 2010, Isle of Eday. Habitat type and code are based on the EUNIS hierarchical classification scheme through visual interpretation (EEA 2004), with presence defined as + for each string transect line (1-3) in both survey sites.

EUNIS habitat type	String number	Location:	
		Device	Control
A 3.11. Kelp with cushion fauna and/ or foliose red seaweeds	1		
	2	+	+
	3	+	+
A 4.112. *Tubularia indivisa* on tide-swept circalittoral rock	1	+	+
	2	+	+
	3		
A 5.444. *Flustra foliacea* and *Hydrallmania falcata* on tide-swept circalittoral mixed sediment	1	+	+
	2	+	+
	3	+	+

5.5. Discussion

Mobile benthic species assemblage: commercial potting assessment

Species diversity

My results are comparable with other marine renewable energy ecological studies, where species diversity estimates were greatest within the renewable energy device deployment survey site (Langhamer 2010). Theory suggests that the addition of new, physical structures to the marine environment can enhance species refuge, feeding and recruitment opportunities, particularly in exposed environments, such as the Isle of Eday (Pardo et al. 2007; Wilhelmsson and Malm 2008). Ecological interactions within naturally exposed marine environments are often limited, and therefore, the addition of such structures can dramatically influence the local assemblage structure (Page et al. 1999). Furthermore, the deployment of new structures within the marine environment can lead to secondary artificial reef habitat scenarios over time. Reef building species, such as mussels or algae, are known to colonise artificial structures across the vertical and horizontal scale, which act as secondary habitats (Langhamer and Wilhelmsson 2009). The process is known to further increase and extend

habitat complexity and species diversity to the adjacent surrounding area over time (Wilhelmsson and Malm 2008). This may therefore explain the observed similarity of diversity estimates between the string lines in the device survey site overall, due to the device originally deployed three years previously. This study however, could not survey the device structure in close proximity (< 200 metres) or at earlier temporal stages (initially post device deployment), due to survey access reasons, to determine this.

Although the deployed device within the Isle of Eday survey site could have some influence on the local benthic communities present, the ecological patterns found are related to the local environmental (extrinsic) and biotic (intrinsic) factors (Martin et al. 2005). This is shown in the variances I observed across the measured temporal scales; with seasonal differences in extrinsic factors such as tidal velocity, wave exposure and water temperature known to heavily affect benthic species diversity (Freeman and Rogers 2003; Zintzen et al. 2008; Smale et al. 2010). During the winter, benthic species are subjected to burial and washouts, with certain species known to actively migrate to deeper waters for refuge from the associated wintering storms and low water temperatures (Sheehy and Prior 2008; Langhamer and Wilhelmsson 2009). In the summer, organisms show enhanced recruitment, growth and settlement intrinsic activities, due to the warmer water temperatures and reduced hydrodynamic events (tidal velocity and wave exposure) (Sheehy and Prior 2008). My diversity results followed this pattern to some degree, with the April 2010 sampling period showing increased diversity estimates from the October 2009 sampling period. Interestingly, the device survey site in the winter October 2010 month also showed the highest diversity overall. This further suggests that the local benthic assemblage could be influenced by the device structure. Other underlying abiotic and biotic processes not recorded within this study may however also contribute to these observations, such as water temperature, recruitment scenarios (Langhamer 2010).

Benthic assemblage composition

The benthic ecological compositions observed within the two survey sites were largely represented by the crustaceans, *N. puber* and *C. pagurus*. Populations of these species are known to dominate the sub-tidal regions of the Orkney Isles, and are associated with rocky outcrops and tidal swept environments (Lee et al. 2006). Habitat preferences by these crustaceans are regardless of sex or age cohorts, and are based on food availability, refuge,

predator avoidance, and actively select structurally complex habitats (Moksnes 2002; Henderson et al. 2005; Pardo et al. 2007; Sheehy and Prior 2008).These species can cope with the strong environmental disturbance regimes represented within the Isle of Eday, due to their heavily armoured exoskeletons and highly mobile characteristics (Sheehy and Prior 2008).

The device site comprised of other mobile species and taxonomic phylums, compared to the control site. This included the echinoderm, *Asterias rubens* and the European lobster, *H. gammarus*, both of which further define tidal swept environments (Connor et al. 2004). *H. gammarus* are nocturnal, shelter dwelling organisms, known to occupy specific habitats such as bedrock, transitional sediments and rocky outcrops habitats (Comeau and Savoie 2002; Galparsoro et al. 2009). Artificial structures can provide additional relief from the strong tides, which are important to *H. gammarus*; by increasing growth, reproduction and feeding opportunities (Galparsoro et al. 2009). Langhamer and Wilhelmsson (2009), also describe that the availability of shelter from predation (natural or anthropogenic) may be a demographic bottleneck for *H. gammarus.* The study outlined that the addition of marine renewable energy structures to the marine environment may act as suitable surrogates of shelter for *H. gammarus*, thus enhancing their restricted populations, at local scale. This scenario may therefore occur with deployed tidal energy devices; due to the presence of this species within the device site and the high similarity of this species' proportion between the device site's string lines (see Appendix 4 Table A.4.2). Further device interaction assessments will be required to determine this, as this study only examined the influence of one operational deployed device across a small spatial area (600 m from device), within the overall site. This could include ecological assessments pre and post new tidal energy device deployments across larger spatial scales and other marine tidal race environments. In addition, other intrinsic and extrinsic factors not addressed within this study may be potentially influencing the notable presence of this species within the device site, such as *H. gammarus* local dispersal routes within the Fall of Warness tidal stream (Galparsoro et al. 2009). I also noted that the caught *H. gammarus* individuals were predominately female (approximately 80%) and that a large proportion of these females were berried. Recent studies suggest that berried female *H. gammarus* individuals travel shorter distances, to maximise egg development and condition, when compared to non-berried females (Comeau and Savoie 2002). This may begin to explain the presence of this species within the device

site, although in-depth surveys regarding reproduction or life stage estimates were not undertaken during the study.

My results follow other past studies, where benthic species and taxonomic composition showed temporal variation (Orlando-Bonaca et al. 2008). Causal factors are commonly linked to a number of biological or environmental processes such as migration, recruitment, water temperature and exposure, described earlier (Page et al. 1999; Lee et al. 2006). Such temporal variations can also affect different segments of a population, in terms of life stage, sex and size, which could be included in future studies (Dunnington et al. 2005). For example, species can aggregate for mating opportunities, or, migrate for spawning or, to avoid osmotic stress during moulting stages (Dunnington et al. 2005; Ungfors 2008).

The observed feeding regimes recorded within these sites can be related to both the deployed device and localised hydrodynamic regimes. Dense proportions of benthic predators are often associated with increased habitat complexity. Past studies describe that predators (such as echinoderms and crabs) are attracted to deployed renewable energy devices, due to the bio-fouling prey species (secondary habitat species) found attached to their foundations (Langhamer 2010). The increased predator proportions I observed within the device site may therefore be due to the presence of the deployed device. Marine predators however, can also alter their prey or habitat choice, to increase feeding opportunities, particularly in areas with few resources (Pardo et al. 2007). This type of opportunistic behaviour is more commonly associated with omnivores, which were predominately observed within both survey sites (Davenport et al. 2011). Overall, my results suggest that species' feeding opportunities within these sites is highly variable, related to the dynamic nature of these environments. This also suggests that where small scale environmental or anthropogenic changes occur, the ecological interaction can be reflected in the type of benthic species feeding regime present.

Benthic habitats: video assessment

Following EUNIS habitat hierarchical classifications, the three benthic habitats identified are regarded as fully marine systems, found in extreme fast flowing wave and tidal stream environments (Connor et al. 2004; Brown and Collier 2008). These habitats are described to support a range of epifaunal species, which show turf and sessile traits or highly mobile characteristics, for which I also found during the commercial potting assessment (Connor et

al. 2004). The observed habitats and associated substrate proportions largely corresponded to their proximity to the Isle of Eday coastline, which suggests that the two survey sites are subject to similar intense hydrodynamic and environmental processes. These processes can influence local species community patterns, nutrient supplies, organic and microbial contents and sediment compositions within the benthic zone (Brown and Collier 2008; Hewitt et al. 2008). Therefore, the impact of the device in terms of the broad-scale benthic habitat classification definitions may be small, in comparison to the overall environmental, biological and chemical processes found within the benthic regions of the Isle of Eday.

Small scale changes may have, however occurred within the immediate area of the device which was not surveyed during this study, or due to the survey methods used. In addition, the EUNIS benthic habitat classification descriptions may not have been detailed enough to define this environment type, the ecological impact from tidal energy device developments. Further habitat classification analysis and study of the benthic habitats within marine tidal race environments will be required to determine this. Additional benthic habitat sub-categories for the EUNIS classification scheme could be developed specifically for offshore renewable energy structures, to aid future habitat and environmental impact assessments. Overall, this highlights the use of benthic habitat assessments as complementary tools to broadly evaluate the benthic ecology within marine tidal race environments and potential impacts from tidal energy devices.

Caveats and recommendations

The methods used in this study are valuable in determining benthic assemblages and habitats where information is poor, but are limited in terms of both fine and wide-scale analysis. Commercial potting methods can also be biased towards the catch-ability of the overall benthic community, whereby they are actively designed to capture crustacean species (Dunnington et al. 2005). Such biases can be reduced by selecting variables including comparable soak times, bait type, repeatability of location deployments and consideration of temporal scales (Dunnington et al. 2005). I implemented these variables into the method design, which was then shown in the range of other species and taxonomic groups within my results. Alternatively, selecting specific crustacean species for additional population studies may also compensate for such biases in the future. This could include the use of mark-recapture methods on species such as *H. gammarus*, which may to give further information

on the environment as a whole, and, the implications of species interactions with tidal energy devices (Comeau and Savoie 2002; Galparsoro et al. 2009; Schmalenbach et al. 2011). The combined use of the benthic species, taxonomic groups and feeding regimes were also useful to highlight different ecological aspects of the two survey site environments. I therefore further recommend the use of these ecological categories for future benthic assemblage assessments.

Habitat video camera assessments can often under represent cryptic species or multi-layered benthic communities, and give a generalisation of the present biota, even in combination with complex bathymetry datasets (Mayfield et al. 1999; Ierodiaconou et al. 2011). Also the habitat classification schemes may well miss the true representation of the benthic community in environments that have been poorly defined; as new habitats or sub-habitats/biotopes in this type of environment may exist (Brown and Collier 2008; Brown et al. 2011). Therefore complementary habitat analysis with scuba-diving video surveys is recommended; with reference to the tidal structure and the immediate surroundings (Mayfield et al. 1999; Maar et al. 2009). This will give a more in-depth review of the local benthic community structure and habitat classification, whilst also outlining the potential for secondary habitat scenarios (Page et al. 1999; Langhamer and Wilhelmsson 2009). In addition, implementing a combined habitat analysis may enable other key ecological assessments, such as the influence of tidal energy devices upon the local biofouling community. Wilhelmsson and Malm (2008) describe how the deployments of offshore wind turbine structures support new biofouling assemblages, influence natural biota on adjacent substrata and attract non-native species, within the local marine environment. These local influences may therefore occur with tidal energy device developments, yet, as few studies exist, the combined habitat analysis could reveal such patterns. This is important, given the current and future scaled developments of these artificial structures within the local and in turn, the wider marine environment.

Overall, recommendations for future study lies with the extension of temporal scales in terms of replicate hauls, seasonal and annual scales, and the spatial scale, first by extending additional string lines ‘within’ these survey sites. This would validate my results and further enhance ecological knowledge of these two survey sites. For example, the string lines only covered approximately 600 m distance, therefore extending the number of string lines (for example, 5 lines coving approximately 1000 m distance from the device) may enhance the results from this study. This may accurately determine the overall influence of the tidal

energy device within the survey site or reveal additional benthic habitats present. Secondly, such studies should be undertaken ‘between ‘other local tidal race environments within the Orkney Isles and beyond, such as the Pentland Firth tidal race. This would enable further assessment of the benthic assemblages and habitats within the Fall of Warness tidal race environment and these environments on the whole. In terms of outlining tidal energy device interactions, these study methods should be implemented where other tidal energy developments occur, such as device deployment, operational activities in the long term and site maintenance activities. This is of particular importance, as small scale surveys (in terms of number of device replicates surveyed) such as this study, could misinterpret the true ecological interaction with these new technologies. Developing studies which include more in-depth temporal, spatial and device replicate scales, would inherently enhance knowledge on the potential anthropogenic interactions with the benthic environment (Langhamer 2010; Shumchenia and King 2010).

Conclusions

This study suggests that the benthic community and habitat structures of the Isle of Eday are heavily influenced by the intense localised hydrodynamic and environmental regimes. Benthic ecosystems are ‘3d’ environments, altered not only by physical attributes such as substrate type or water depth, but also by the overlying water column, which supplies food, gametes, nutrients and new recruits (Brown et al. 2011). Such factors can alter benthic community structures at the temporal scale, with species composition and functionality in the Isle of Eday shown to change across seasonal scales.

Overall, the tidal energy device has some ecological presence on the benthic community within the EMEC test site region, which is represented in the measured differences of the benthic patterns observed between the survey sites (Langhamer et al. 2009). Marine artificial structures can alter the distribution and abundance of benthic species at the local scale by modifying physical habitat, tidal regimes and local infaunal and epifaunal benthic communities (Pardo et al. 2007). These interactions could be species, taxonomic specific, related to the life history of the individual species or habitat specific, whereby interactions depend upon the existing seabed type. Our wider understanding of how these devices interact in the ecological sense are currently based on computerised model scenarios, or fieldwork orientated round one test or commercial device in a specific site. In the long term, further

replication in terms of sampling more devices is suggested as more deployments are carried out, to determine the true nature of the interaction.

The survey methods were useful with context to the aims of the study; to provide ecological information on the present benthic community structure and the potential interactions with a tidal energy device. These strategies have some caveats, namely the potential biasness for crustacean species and the broadness of the video habitat classifications, which may miss the true picture of the environment or device interaction. I therefore recommend additional methods to complement this study, such as dive transects or crustacean mark-recapture assessments that could be implemented across further temporal and spatial scales. At the wider level, studies such as these could be combined with hydrodynamic and tidal energy extraction models and applied to new deployment activities, to begin to understand the potential interactions with large scale tidal energy developments in the long term (Brown et al. 2011; Neill et al. 2012).

Chapter 6. Ecological variation of intertidal rocky-shore communities within a marine tidal energy development site.

6.1. Abstract

Intertidal macroalgae and faunal species are important ecological units within the marine environment and are useful biological monitoring tools to assess anthropogenic environmental impacts. Tidal energy development is a new anthropogenic activity within the marine environment, with few quantitative environmental impact assessments conducted within the regions of the intertidal system. I examined intertidal ecological macroalgae and faunal components within the EMEC tidal energy test development site, Orkney, to advance current baseline records and outline potential environmental impacts. I used field based sampling techniques to determine macroalgae and faunal species biodiversity, composition and life history traits, across different shoreline height levels and temporal sampling periods, from 2009 to 2010.

My results indicate macroalgae biodiversity and composition differed between the development site and the control coastlines. This was due to the added presence of species such as *Codium fragile* subsp. *fragile* identified within the development site coastline, which is an indicator of coastal anthropogenic activities. Faunal species biodiversity and composition showed weak differences between the two coastlines, due to the dominating presence of molluscan and cnidarian species. I also identified that macroalgae and faunal species diversity and composition showed minor changes across the measured temporal sampling periods. These results suggest that macroalgae and faunal species exhibit life history traits and ecological patterns associated to the dynamic nature of this exposed intertidal environment, with possible small-scale reflections of anthropogenic impact from tidal energy development activities.

6.2. Introduction

The development of offshore marine renewable tidal energy production schemes within the British Isles territorial waters has grown dramatically since the last ten years, with increased interest expected within the next two decades (Allan et al. 2008). The British Isles currently sustains a number of small scale tidal energy development schemes (1 – 10 tidal energy device deployments) and future large scale (100 + tidal energy device deployments) commercial array design proposals (Neill et al. 2009; Bahaj 2011; Shields et al. 2011). Existing small schemes include tidal energy test development sites, designed for manufacturers to trial single commercial or test-scaled tidal energy devices under real environmental conditions within offshore marine tidal race environments (Norris and Droniou 2007; Shields et al. 2011). These development sites are used for a number of tidal energy device research activities, such as vessel visits for resource assessments, cable laying placements, pile-driving, device deployments and device energy generation trials.

The intertidal coastlines within, or adjacent to tidal energy development sites can be described as a dynamic environment, with ecological communities comprising of a mosaic of macroalgae and faunal species assemblages (Wells et al. 2007; Juanes et al. 2008). This is due to the strong hydrodynamic regimes found within these intertidal environments, influencing the communities present (Shields et al. 2011). A number of past studies suggest that the communities within these environments comprise of few species, and show biological traits or behaviours adapted to these extreme environmental conditions (Hiscock 1983; Shields et al. 2011). For example, Hiscock (1983) outlines that when compared to non-tidal race environments, species show high mechanical stress (such as macroalgae encrusting body forms) or rapid feeding regimes traits (faunal scavenging or filter feeding types), which enable them to cope with the strong hydrodynamics present.

Macroalgae are regarded as a vital group within intertidal regions, and are directly affected by a range of physical, chemical and biological factors, due to their sessile characteristics (Orfanidis et al. 2001). These extrinsic and intrinsic factors also influence faunal species assemblages, with species exhibiting a range of life history traits and habitat preferences associated with these exposed marine intertidal environments (Schiel 2004; Menge et al. 2005; Guerra-García et al. 2011). Both macroalgae and faunal assemblages are ecologically important, shown through their repetitive use for scientific investigation and aid in defining the ecological status of coastal water bodies following environmental impacts, such as oil pollution (Underwood 2000; Guinda et al. 2008).

At present, in-depth ecological, quantitative baseline knowledge of the intertidal regions within marine tidal race environments is limited, particularly within tidal energy test development sites (Shields et al. 2011). Despite preliminary past studies and records, predicting the ecology within these environments is difficult (Hiscock 1983). This is due to the complex interaction of different hydrodynamic regimes (wave exposure, tidal velocity, and turbidity), other environmental variables (substrate type, depth, water temperature) and in turn, the ecology present. In addition, the environmental impacts associated with these anthropogenic activities are also unknown, due to tidal energy technology currently being at an early research phase, and because the overall range of activities at development sites is quite varied (Gill 2005). Therefore, investigating key ecological communities such as intertidal macroalgae and faunal species assemblage compositions can give valuable information into the ecology of these environments and also begin to identify the potential environmental impacts associated with this type of new anthropogenic activity. This is of particular importance in the long term due to the potential wide scale and increased scope for the development of tidal energy schemes across the British Isles (Bahaj 2011; Shields et al. 2011).

The aim of this chapter is to investigate the intertidal macroalgae and faunal species assemblages within an existing tidal energy test development site. This intertidal community composition assessment was conducted within the intertidal coastlines of the European Marine Energy Centre's (EMEC) tidal energy test development site, Orkney Isles. This site has been managed by EMEC since 2001; to enable offshore renewable tidal energy device manufacturers to develop and trial tidal energy device performance (Norris and Droniou 2007).

Here, I focus upon examining species diversity, assemblage composition and functional form type. These parameters have been used in a variety of studies to define marine ecological patterns, their relationship to changing environmental gradients (wave exposure, tidal velocity, depth) and impacts from anthropogenic activities such as other renewable energy development (wave energy device deployment) and coastal development (coastal defence development) (Martin et al. 2005; Langhamer et al. 2010). This research will also entail comparing these assemblages across different shoreline heights and seasonal temporal scales. The use of these measured ecological scales gives a wider-perspective of the ecological variation that can occur within the intertidal environment, and in turn provide greater accuracy of the potential associated environmental impacts (Rindi and Guiry 2004; Bates et

al. 2007; Langhamer 2010). For this chapter, I test the hypothesis that the tidal energy test site will influence the intertidal ecology across the measured spatial scales (coastlines and shoreline heights). I also test the hypothesis that this anthropogenic activity will influence the intertidal ecology regardless of the measured temporal scales.

6.3. Methods

Study area

The study area was located across two intertidal coastlines of the Isle of Eday, which is part of the Orkney Isles, Scotland. This comprised of the coastline within the EMEC tidal energy research test facility site and Fall of Warness tidal race environment (Figure 6.1a; regarded as the device coastline (DC)), situated along the South-West coastline of Eday. I also selected a second coastline to act as a control for ecological comparisons (Figure 6.1a; regarded as the control coastline (CC)), which was located along the Eastern coastline of Eday. I selected this coastline to act as the control based on its similarity to the device coastline, in terms of physical and biological characteristics, such as; water flow conditions, wave exposure, coastline profile (rocky shoreline) and substrate properties. Intertidal substrates found within both coastlines comprise of a mixture of exposed bedrock, boulders and cobbles, with coarse gravels and sands (Aquatera Ltd 2008).

During this study, the device coastline comprised of one permanently deployed tidal energy test device, designed by OpenHydro Ltd. This device is located on the most northerly sub-sea cable position. In addition, the site was frequently visited by OpenHydro Ltd, EMEC and other offshore research vessels during the study for device maintenance and marine renewable energy research trial activities (personal observation). Within the control coastline I did not observe any tidal energy device deployment or research activities.

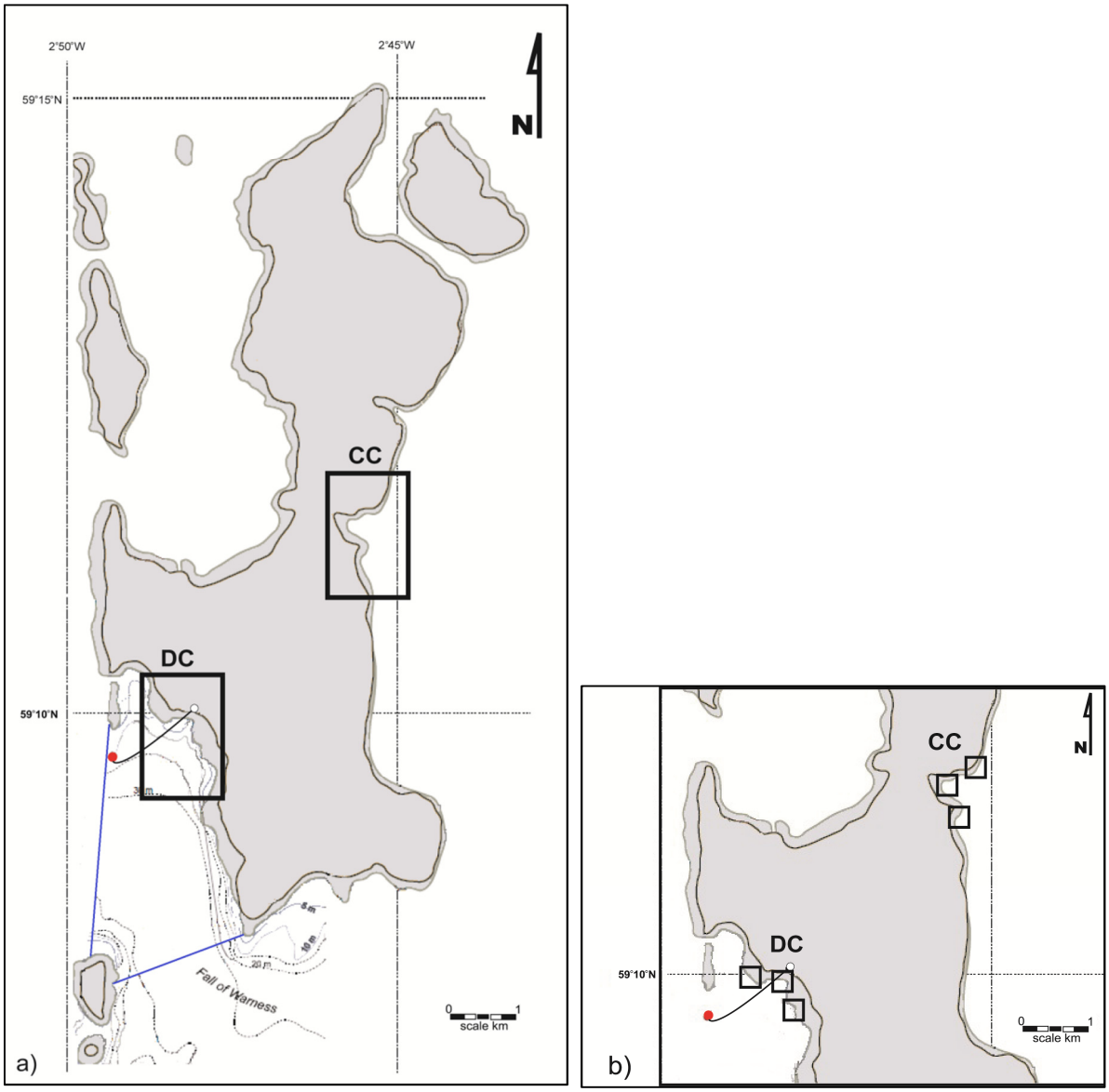


Figure 6.1. Location of the intertidal community composition study, 2009 – 2010, Isle of Eday (adapted from Aurora 2005). a) The device coastline study area (DC) within the EMEC tidal energy test development site and the control coastline (CC) study area. Extent of the EMEC tidal energy test site is within the blue lines. The red circle and joined black line shows the position of the deployed OpenHydro Ltd test device and sub-sea cable route. b) The location of the three survey bay sites selected within the device and control coastlines.

Within both coastlines, I selected three intertidal survey bay sites, which were approximately 500 metre distances away from each other, for ecological replication efforts (Figure 6.1b; Appendix 5 Table A.5.1 for bay positions). I then split each survey bay site into two sections based on their tidal shoreline height and described them as upper and lower shoreline heights (Skalski et al. 2001). These shoreline heights ranged between the mean high water level of spring tides (MHWS) and the mean low water level of spring tides (MLWS). They were selected from the start of the upper shoreline section (MHWS) and calculating the mid-point between the high and low tide points. The area of each shoreline height across all survey bay sites was approximately 250 metres in length and 100 metres in height. These shoreline heights were based on tidal range information from Chart Datum (Admiralty Nautical Charts 2007) and an earlier ecological preliminary pilot study, which I conducted in September 2009 (ecological comparisons between shoreline heights). All shoreline visits were standardised to start approximately two hours before the local low tide time, with surveys conducted with the falling/ebbing tide (Nedwell et al. 2002; Horton and Murray 2006). These visits were undertaken during each sampling month's 'spring' tide periods.

Intertidal community composition assessment

The intertidal community composition assessment was conducted within both coastlines, across four sampling periods: October 2009, April 2010, July 2010 and October 2010. These were selected to represent seasonal temporal periods (winter, spring, summer), which are used in similar studies (Menge et al. 2005).

I used a random stratified sampling technique, where I fixed four 100 metre transect lines perpendicular to the coastline, within both shoreline heights (upper; mid), across all survey bay sites (1-3), for both coastlines (device; control). This was repeated across the four temporal seasonal sampling periods. The starting position of each transect was located at the top of each shoreline height section (upper: MHWS; lower: top of mid-point), and placed approximately 50 metres away from each other (Figure 6.2). These positions were based on the overall area of each survey bay site, taken from Chart Datum (Admiralty Nautical Charts 2007). The start and end position of each fixed transect was recorded by dGPS (dGPS, with accuracy to 3.65 metres). These fixed transect positions were then repeatedly used for each temporal seasonal sampling period.

Along each 100 metre transect, I marked sampling station points every 4 metres and numbered them 1 to 26. Ten gridded 0.5 m² quadrats were then placed at randomly selected station points along each transect, for the community composition assessment. The quadrat station numbers (1-26) were randomly selected using a random number generation application in R (R Development Core Team 2010). These station numbers were randomly generated for each transect throughout the study.

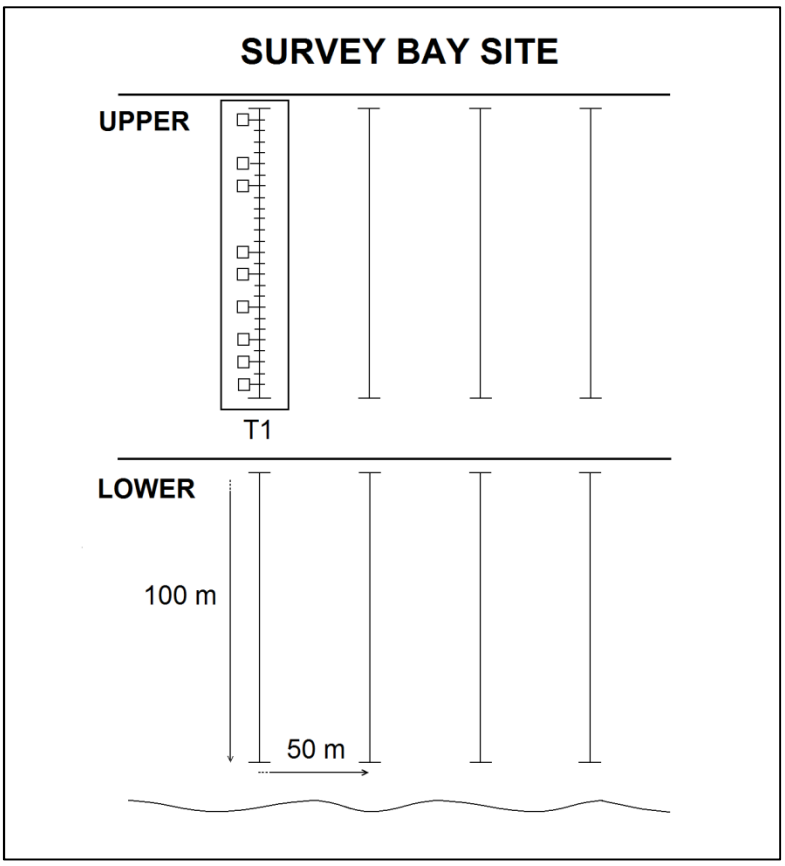


Figure 6.2. Schematic diagram of the 2009 - 2010 intertidal community compositional study sampling method within one survey bay site. The survey bay site is split into upper and lower shoreline heights, with four 100 metre transect lines running perpendicular to the shoreline within each shoreline height. On each transect (example shown as T1), ten 0.5 m^2 quadrats (shown as squares) were placed randomly along 26 pre-defined stations with a separation of 4 m along each transect. This survey layout is replicated within each shoreline height, survey bay site, and temporal seasonal sampling period.

Intertidal species community composition was identified through estimation of macroalgae species percentage cover (%) and total faunal species (singleton) abundance counts (n) within each randomly placed quadrat. Each quadrat was photographed, with the position recorded by dGPS. All organisms were identified to at least species level in the field. I also collected particular macroalgae (Genus: *Porphyra*) mollusc species (Genus: *Gibbula*) for further taxonomic identification by microscopic analysis and independent identification verification (Hayward and Ryland 1995; Hayward et al. 2001; Bunker et al. 2010; Fish and Fish 2011; Guiry and Guiry 2012).

Data analysis

Data were analysed using the R statistical software (R Development Core Team 2010), with all multivariate exploratory routines using the Primer v6 software package (Clarke and Gorley 2006). I assessed intertidal community composition by averaging macroalgae species percentage cover and faunal species abundance counts across the four transects, within all coastline survey bay sites, for both shoreline heights. This was undertaken for each temporal seasonal sampling period.

To examine species diversity differences between the two coastlines and four temporal seasonal sampling periods, I generated macroalgae and faunal species diversity estimates for all coastline survey bay sites within both shoreline heights using the Primer v6 DIVERSE routine (total number of data points for each shoreline height = 24). This included: species richness (*S*), Shannon diversity (H', $\log_e$) and Pielou's evenness (J'). I then used generalised linear models (GLM) to look for differences between these diversity measures as a function of coastline and temporal seasonal sampling period, within each shoreline height. The species richness (*S*) GLMs used a Poisson error distribution with a log link and, where overdispersion occurred, the quasi-poisson error distribution was used. The Shannon-Wiener (H', $\log_e$) and Pielou's evenness (J') diversity measures used a Gaussian error distribution. I first generated GLM maximal models, which included all interactions between the coastline and temporal seasonal sampling period. These fully factorial models were then simplified using deletion of non-significant ($p > 0.05$) terms (using ANOVA with appropriate F or Chi tests), to produce minimal adequate models (Crawley 2007). All minimal adequate models were then checked visually for non-normality using Q-Q normality plots, and transformed by logs, where necessary.

I then compared macroalgae and faunal species, taxonomy and functional form composition between the coastlines and temporal seasonal sampling periods, for both shoreline heights using multivariate analyses (Table 6.1). For macroalgae taxonomic types, I classified species to the taxonomic level of division, using taxonomic keys and species on-line databases (Hayward and Ryland 1995; Appeltans et al. 2012; Guiry and Guiry 2012). Macroalgae functional form was based on morphological forms, using the Steneck and Dethier (1994) classification scheme. Faunal species were classified to the phylum level for taxonomic analyses, with functional form based on their feeding regime types derived from the BIOTIC database (Hayward and Ryland 1995; MarLIN 2006; Appeltans et al. 2012). Prior to these multivariate analyses, I fourth root transformed the macroalgae percentage cover and faunal species count datasets, to weight the contributions of common and rare species (Clarke and Gorley 2006). I selected this transformation to view these ecological communities in more depth (Clarke and Warwick 2001).

Table 6.1. Intertidal macroalgae and faunal species, taxonomic and functional forms used for multivariate analyses during the study. Macroalgae taxonomy analyses are based on the division rank, with faunal species taxonomic analyses based on the phylum rank. Macroalgae Ochrophyta phylum species are regarded as Phaeophyta phylum for this assessment, for ease of interpretation. Macroalgae morphological functional form classification follows the Steneck and Dethier 1994, classification system. Faunal species feeding regime functional form follows the BIOTIC on-line species database (MarLIN 2006).

Species	Taxonomic categories	Functional form
Macroalgae	Chlorophyta	*Morphology*
	Phaeophyta	Filamentous algae
	Rhodophyta	Filamentous (2)
		Foliose single
		Foliose corticated
		Corticated macrophyte
		Leathery macrophyte
		Articulated calcareous
		Crustose algae
Fauna	Cnidaria	*Feeding regime*
	Echinodermata	Epifaunal herbivore
	Mollusca	Omnivore
		Predator
		Suspension feeder

To identify significant differences of the composition of species, taxonomic and functional forms between the two coastlines and four temporal seasonal sampling periods, I used one-way analysis of similarities. This analysis was run using the Primer v6 ANOSIM routine, which is based on Bray-Curtis distances between samples and compares the ranks of distance values within and between samples, which are grouped as a factor. The analysis produces a significance level and a global absolute statistic R, which ranges from -1 (large differences within groups) to 1 (large differences between groups), using random permutations of the grouping factor to assess significance (Langhamer et al. 2009). I first ran this routine across all datasets to identify compositional differences between the two coastlines, for each shoreline height. I then re-ran this routine to outline temporal compositional differences within the two coastlines separately, for each shoreline height. To identify which species, taxon and functional form compositions were significantly responsible for the biotic patterns shown by these ANOSIM tests, I used the one-way Primer v6 similarity percentage SIMPER routine. This routine generates the contribution of each species or group within a sample as a percentage, and is based on the Bray-Curtis distances, to assess the 'closeness' of samples within a group (Clarke and Warwick 2001; Clarke and Gorley 2006). The contribution similarity percentage for macroalgae and faunal species composition, taxonomic and functional form samples were based on 100 % cut off of the total percentage contributions, due to the small number of recorded variables (Clarke and Warwick 2001).

I finally explored the overall structure of macroalgae and faunal species compositions (across all coastlines and temporal seasonal sampling period samples) for each shoreline height visually, through cluster analysis dendograms. Hierarchical cluster dendograms were generated from the resulting Bray-Curtis similarity resemblance matrices. These were generated in the Primer v6 CLUSTER routine, with the coastline and temporal seasonal sampling periods selected as factors, and the group average link selected.

6.4. Results

I recorded a total number of 20 intertidal macroalgae species and 11 faunal species across the study (Appendix 1 Table A.1 for species lists). Macroalgae comprised of 5 Chlorophyta, 7 Phaeophyta and 8 Rhodophyta species, which represented a range of morphological forms. Faunal species comprised of 1 Cnidaria, 1 Echinodermata and 9 Mollusca species, which represented epifaunal herbivore and predator feeding regimes.

Macroalgae diversity

In the upper shoreline height, macroalgae species richness (*S*) (mean *S*: device = 10.08; control = 6.91) and Shannon-Wiener diversity estimates (mean H': device = 2.189; control = 1.817) were significantly larger within the device coastline, compared to the control coastline (Table 6.2; Figure 6.3a-c). The GLM results showed highly similar Pielou's evenness (J') diversity estimates between the device and control coastlines (mean J': device = 0.959; control = 0.951). Overall, I found similar macroalgae diversity estimates across the four temporal seasonal sampling periods, within both coastlines.

In the lower shoreline height, I also found significantly greater estimates of macroalgae species richness (mean *S*: device = 11.66; control = 9.50) and Shannon-Wiener diversity (mean H': device = 2.339; control = 2.124) within the device coastline (Table 6.2; Figure 6.3d-f). Pielou's evenness values were highly comparable between the device and control coastlines overall (mean J': device = 0.955; control = 0.954). Within this shoreline height, I also found macroalgae diversity was similar across the four temporal seasonal sampling periods, within both coastlines.

Table 6.2. Generalised linear minimal adequate models (MAM) for intertidal macroalgae species diversity measures within the upper and lower shoreline heights, from 2009 - 2010, Isle of Eday. Species diversity measures include Species richness (*S*), Shannon-Wiener (H', $\log_e$) and Pielou's evenness (J') diversity. F represents the overall F model value; P represents the model P value; DF represents the degrees of freedom; n.s. represents no significant variables (n = 24). The upper shoreline Pielou's evenness model is based on a log transformed and the species richness models are based on a quasi-poisson distribution.

Shoreline height	Model explanatory variables	F value	P	DF
Upper	**Species richness (*S*)**			
	Intercept			
	Coastline	15.00	< 0.001	22
	Shannon-Wiener diversity (H')			
	Intercept			
	Coastline	16.49	< 0.001	22
	Pielou's evenness (J')	n.s		
Lower	**Species richness (*S*)**			
	Intercept			
	Coastline	7.22	0.013	22
	Shannon-Wiener diversity (H')			
	Intercept			
	Coastline	5.94	0.02	22
	Pielou's evenness (J')	n.s		

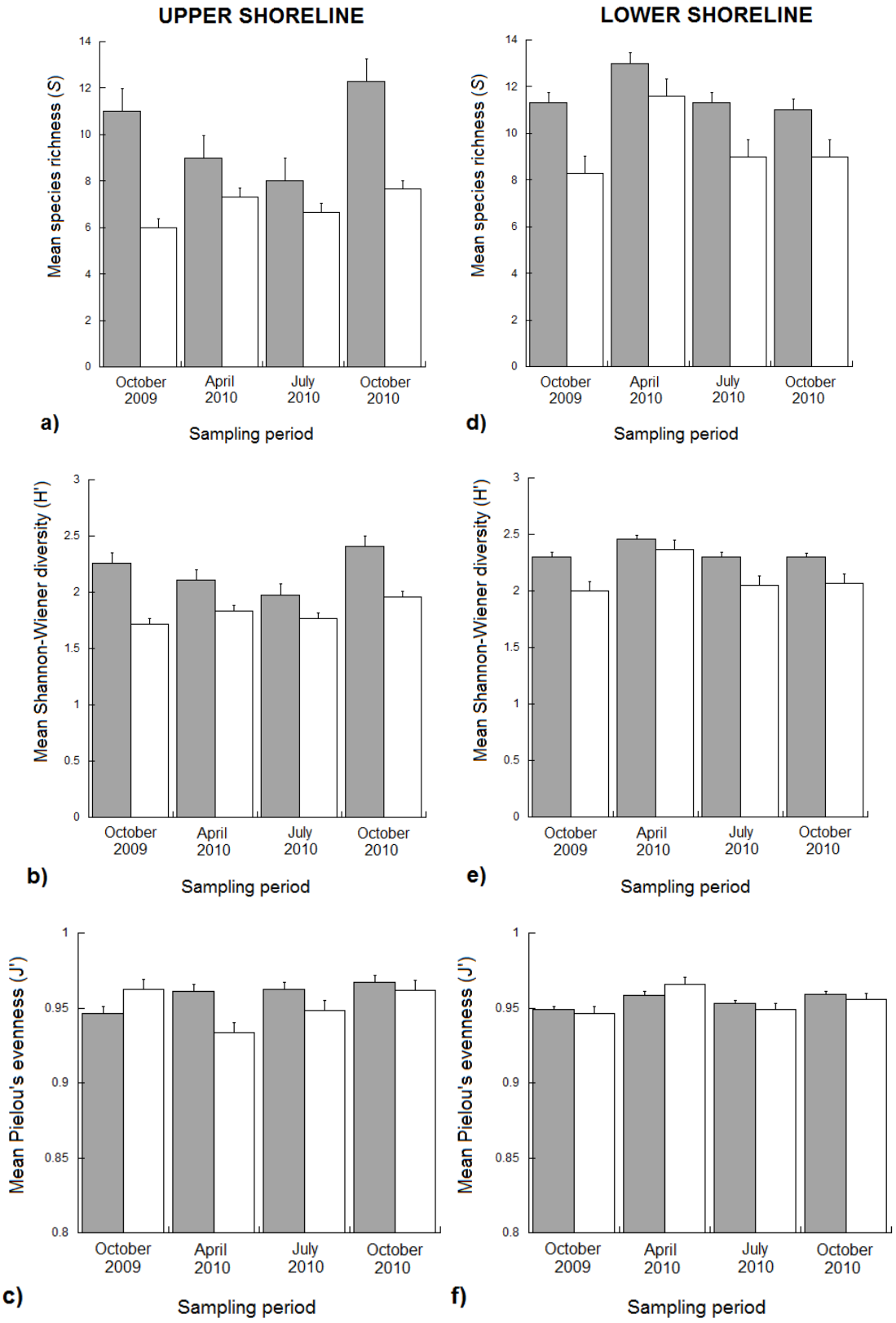


Figure 6.3. Mean intertidal macroalgae diversity measures in the upper a - c) and lower d –f) shoreline heights, within the device and control coastlines, from 2009 - 2010, Isle of Eday. Grey bars represent the device coastline; white bars represent the control (± 1 SEM).

Macroalgae composition

The ANOSIM results showed that macroalgae species composition in the upper shoreline was significantly different between the two coastlines (Table 6.3). The SIMPER results revealed that the device coastline comprised of additional species, including *Codium fragile* subsp. *fragile* (Table 6.4). The cluster analysis and ANOSIM results however, showed that macroalgae species composition was largely similar between the four temporal seasonal sampling periods, for both coastlines (Figure 6.4a; Appendix 5 Table A.5.2 for temporal SIMPER results). At the taxonomic division level, macroalgae composition was highly comparable between the two coastlines and across the temporal seasonal sampling periods. I also found that macroalgae morphological forms were significantly different between the two coastlines, shown by the ANOSIM results. The SIMPER results identified that the device coastline comprised of larger proportions of the crustose form and lower proportions of the leathery macrophyte form. These ANOSIM results also showed that macroalgae morphological form was comparable across the temporal sampling periods, within both coastlines.

Within the lower shoreline, the ANOSIM results also showed significantly different macroalgae species composition between the two coastlines (Table 6.3). The SIMPER results showed that the control coastline comprised of greater proportions of *Ascophyllum nodosum* (Table 6.4). The cluster analysis and ANOSIM results both showed that species composition was highly similar across the four temporal seasonal sampling periods (Figure 6.4b and also Appendix 5 Table A.5.3 for temporal SIMPER results). Macroalgae taxonomic divisions within this shoreline were also comparable between the two coastlines, from the ANOSIM results. From these results, I also found that the composition of these taxonomic division were similar across the four temporal seasonal sampling periods. Macroalgae morphological form showed some differences between the two coastlines, identified by the ANOSIM results. The SIMPER results outlined that the device coastline comprised of greater proportions of foliose and filamentous forms, compared to the control coastline. Across the temporal seasonal sampling periods, I found that these morphological forms were comparable within both coastlines overall.

Table 6.3. Results of the ANOSIM tests of intertidal macroalgae species composition a), taxonomic division composition b), and morphological form composition c), between the two coastlines (device; control) and four temporal seasonal sampling periods, within the upper and lower shoreline heights, from 2009 - 2010, Isle of Eday. Tests are based on one-way ANOSIM analyses across each bay site (1-3) between each coastline and temporal seasonal sampling periods (October 2009; April 2010; July 2010; October 2010), separately for each shoreline height. All results were generated on Bray-Curtis similarity resemblance matrices.

	Global values			
	Upper shoreline		**Lower shoreline**	
	R	***P***	**R**	***P***
a) Species				
Coastline	0.325	0.002	0.209	0.006
Temporal: Device Coastline	-0.07	0.70	0.043	0.35
Temporal: Control Coastline	0.031	0.42	0.083	0.28
b) Taxon				
Coastline	-0.015	0.61	0.004	0.35
Temporal: Device Coastline	0.05	0.26	0.123	0.15
Temporal: Control Coastline	-0.13	0.93	-0.06	0.60
c) Morphological form				
Coastline	0.162	0.01	0.09	0.04
Temporal: Device Coastline	-0.07	0.67	0.136	0.14
Temporal: Control Coastline	-0.012	0.51	0.02	0.39

Table 6.4. Intertidal macroalgae species a), taxonomic division b), and morphological form composition c), SIMPER percentage contributions (%) between coastlines (device; control) in upper and lower shoreline heights, from 2009 - 2010, Isle of Eday. Percentages are generated from one-way routines, based on Bray-Curtis average dissimilarity contributions, with bold indicating ≥ 20 %, and average within-group similarities shown.

	Upper shoreline		**Lower shoreline**	
	Device	**Control**	**Device**	**Control**
a) Species				
Chlorophyta				
Blidingia minima		0.20		
Cladophora rupestris	8.43	1.72	8.86	3.86
Codium fragile subsp *fragile*	2.36		2.06	
Ulva intestinalis	11	11.78	6.06	6.68
Ulva lactuca	0.53	0.98	3.14	1.22
Phaeophyta				
Ascophyllum nodosum	18.65	**25.07**	5.06	**20.42**
Fucus serratus	6.29	1.10	**22.23**	**22.48**
Fucus spiralis	14.30	**27.94**	0.40	
Fucus vesiculosus	**20.51**	**20.27**	17.97	17.57
Himanthalia elongata			5.05	1.86
Laminaria digitata			8.47	2.55
Pelvetia canaliculata	5.53	2.87		
Rhodophyta				
Chondrus crispus			1.97	1.72
Corallina officinalis	0.28		0.89	
Lithophyllum spp	4.81	0.14	6.46	7.28
Osmundea pinnatifida	0.93		6.54	5.09
Porphyra umbilicalis	0.53			
Rhodothamniella floridula			0.12	
Vertebrata lanosa	5.84	7.93	4.71	9.27
Average similarity	71.68	73.22	69.15	75.67
b) Taxon				
Chlorophyta	**26.02**	**22.10**	**21.61**	18.16
Phaeophyta	**55.46**	**63.92**	**54.89**	**57.53**
Rhodophyta	18.52	13.98	**23.50**	**24.31**
Average similarity	89.20	87.36	91.28	91.87
c) Morphological form				
Filamentous	**24.20**	**21.55**	17.13	12.37
Filamentous (2)	10.90	13.41	7.79	15.60
Foliose single	1.98	1.66	6.22	2.04
Corticated macrophyte	1.78		12.84	8.57
Leathery macrophyte	**52.05**	**63.15**	**44.71**	**49.12**
Articulated calcareous	0.51		1.35	
Crustose	8.58	0.23	9.95	12.30
Average similarity	77.92	82.46	81.30	81.11

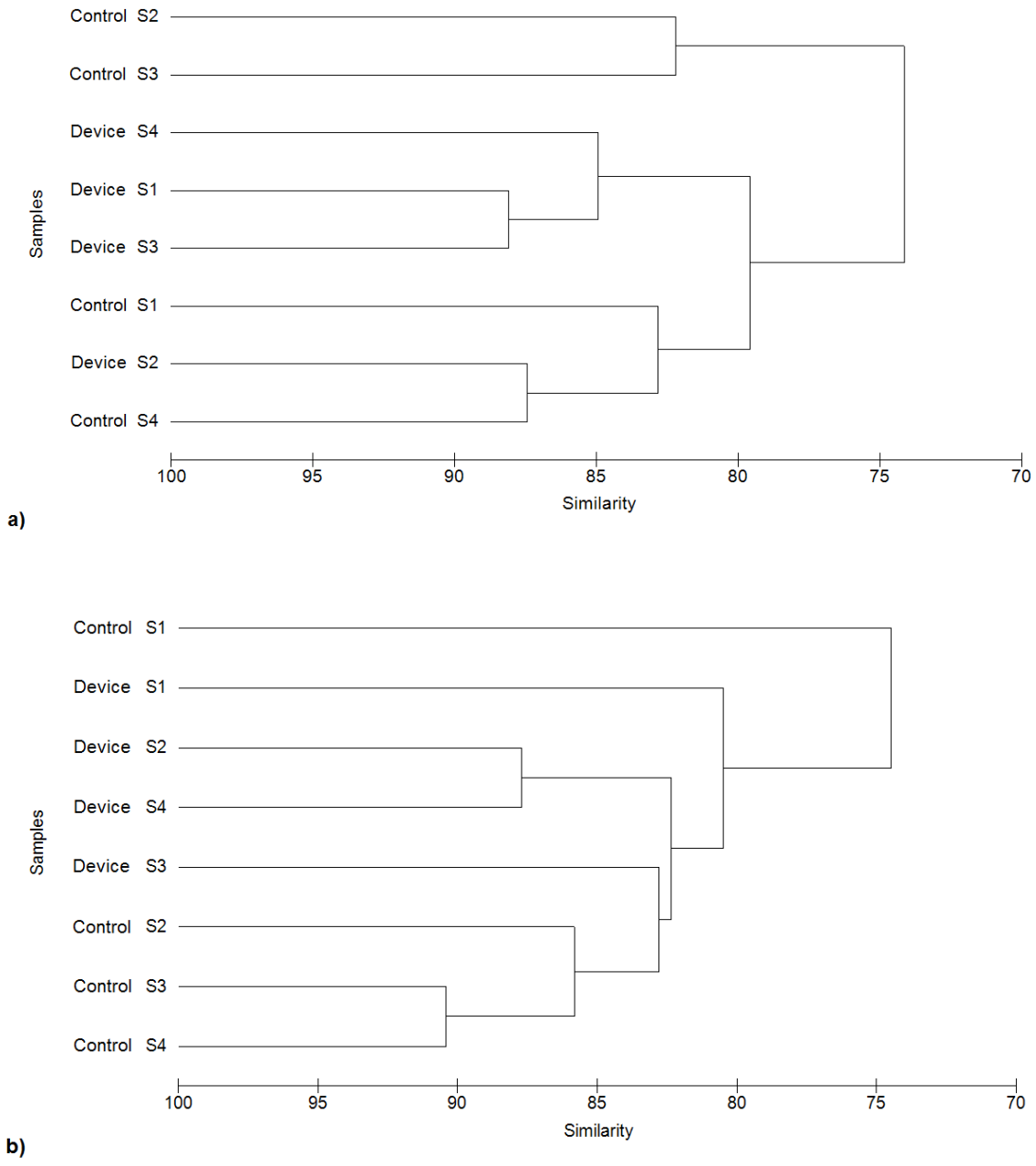


Figure 6.4. Hierarchical cluster analysis dendograms of intertidal macroalgae species composition across the device and control coastlines and four temporal seasonal sampling periods within the upper a) and lower shoreline heights b) from 2009 - 2010, Isle of Eday. Lettering corresponds to the temporal seasonal sampling periods; October 2009 = S1, April 2010 = S2, July 2010 = S3, October 2010 = S4. Cluster analysis is based on the Bray-Curtis similarity resemblance matrix, with group average linking.

Faunal species diversity

I observed similar faunal species richness (*S*) estimates across the device and control coastlines (mean *S*: device = 3.83; control = 2.66), which was also shown in the Shannon-Wiener diversity (H') measures (mean H': device = 1.20; control = 1.01) (Table 6.5; Figure 6.5a-c). The Pielou's evenness (J') measure outlined significant differences between the coastlines, with the control coastline comprising of a greater spread of faunal species (mean J': device = 0.959; control = 0.971). Across the temporal seasonal sampling periods, I identified comparable, minor fluctuations of faunal species richness and Shannon-Wiener diversity. Pielou's evenness differed across the temporal seasonal sampling periods, with decreased evenness found in April 2010, and October 2010 overall.

In the lower shoreline, I also observed similar faunal species richness (mean *S*: device = 5.25; control = 4.58) and Shannon-Wiener diversity estimates (mean H': device = 1.56; control = 1.44; Table 6.5; Figure 6.5d-f). I also found significantly lower Pielou's evenness within the device coastline compared to the control coastline (mean J': device = 0.955; control = 0.970). Across the four temporal seasonal sampling periods, all diversity measures showed similar estimates within both coastlines.

Table 6.5. Generalised linear minimal adequate models (MAM) for intertidal faunal species diversity measures within the upper and lower shoreline heights, from 2009 - 2010, Isle of Eday. Species diversity measures include species richness (*S*), Shannon-Wiener (H', $\log_e$) and Pielou's evenness (J') diversity. F represents the overall F model value; P represents the model P value; DF represents the degrees of freedom; n.s. represents no significant variables (n = 24). The species richness models are based on a quasi-poisson distribution.

Shoreline Height	Model explanatory variable	F value	P	DF
Upper	**Species richness (*S*)**	n.s		
	Shannon-Wiener diversity (H')	n.s		
	Pielou's evenness (J')			
	Intercept			
	Coastline	6.18	0.02	22
	Temporal sampling period	3.29	0.045	20
Lower	**Species richness (*S*)**	n.s		
	Shannon-Wiener diversity (H')	n.s		
	Pielou's evenness (J')			
	Intercept			
	Coastline	6.25	0.02	22

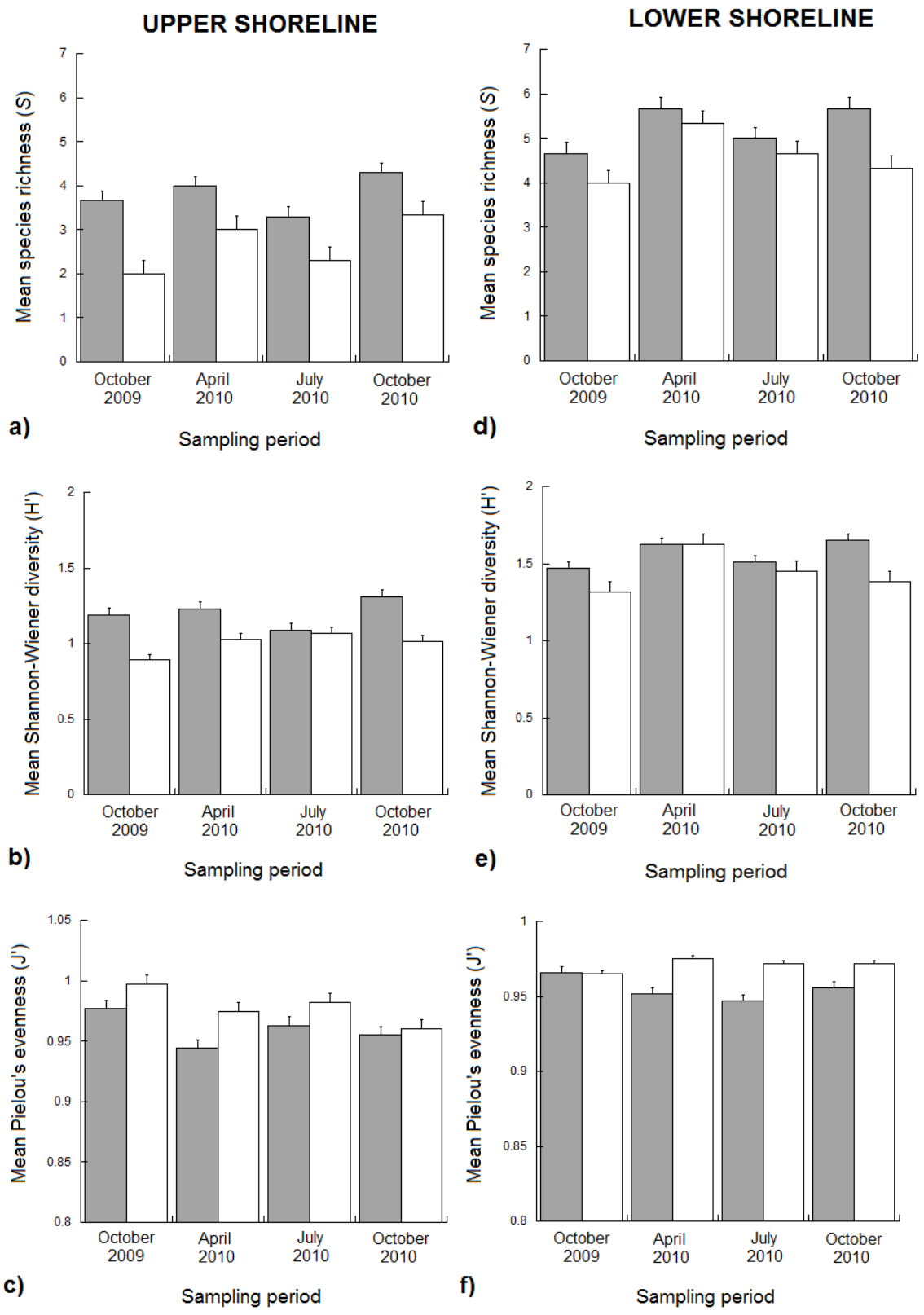


Figure 6.5. Mean intertidal faunal diversity measures in the upper a - c) and lower d - f) shoreline heights within the device and control coastlines, from 2009 - 2010, Isle of Eday. Grey bars represent the device, white bars represent control coastline (± 1 SEM).

Faunal species composition

The two coastlines within the upper shoreline comprised of different faunal species compositions, shown by the ANOSIM results (Table 6.6). From the SIMPER results, I found this was due to the presence of *Actinia equina* and larger proportions of *Patella vulgata* within the device coastline (Table 6.7). The ANOSIM results also showed that both coastlines consisted of different species compositions across the four temporal seasonal sampling periods. The cluster analysis outlined the control coastline during October 2009 contributed to the most species composition dissimilarity across the two coastlines and sampling periods (Figure 6.6a; see Appendix 5 Table A.5.4 for temporal SIMPER results). The ANOSIM taxonomic results outlined significantly different taxonomic compositions between the two coastlines. The SIMPER results, identified that the control coastline predominately comprising of mollusc species, compared to the device coastline. The ANOSIM results also outlined that both coastlines comprised of similar compositions across the four temporal seasonal sampling periods. The feeding regimes of the recorded faunal species differed between the two coastlines from the ANOSIM results, with the SIMPER results outlining fewer predator proportions within the control coastline. I found that these feeding regime types within both coastlines were also comparable across the temporal seasonal sampling periods, shown by the ANOSIM results.

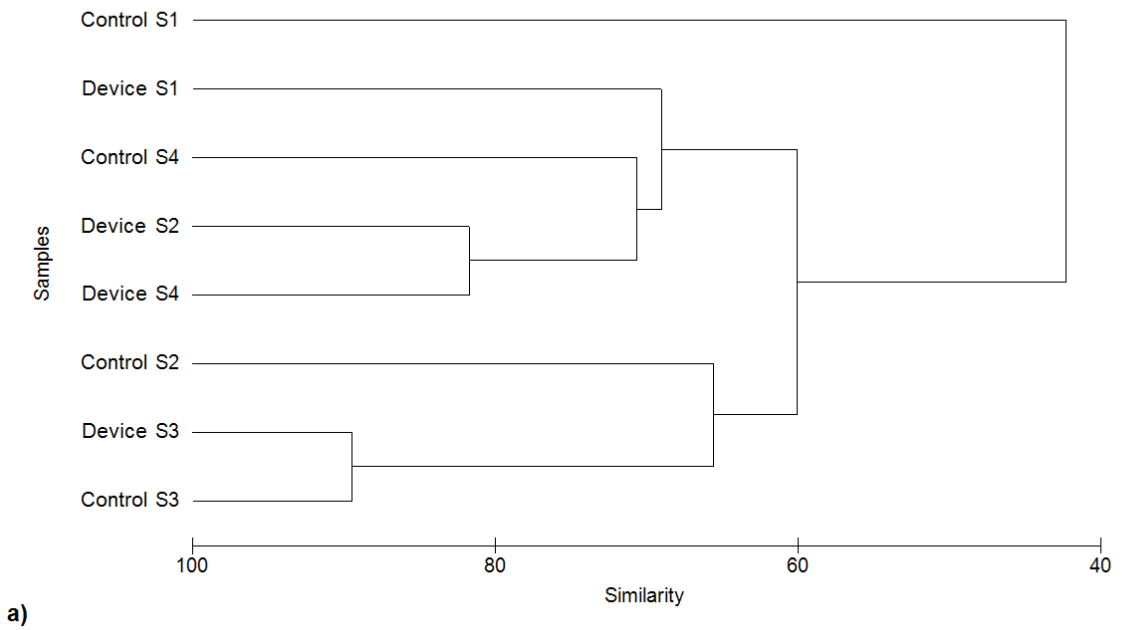
Faunal species composition within the lower shoreline was similar between the two coastlines, shown by the ANOSIM and SIMPER results (Table 6.6; Table 6.7). The cluster analysis showed that species composition was largely comparable between the two coastlines and temporal seasonal sampling periods (Figure 6.6b). The ANOSIM results also showed minor differences between the sampling periods within the device coastline (Appendix 5 Table A.5.5 for temporal SIMPER results). From the ANOSIM results, I also identified that the two coastlines comprised of different taxonomic group compositions. I found that this was due to larger proportions of molluscan species within the control coastline, shown by the SIMPER results (Table 6.7). The ANOSIM results showed that the composition of these taxonomic groups within both coastlines were similar across the four temporal seasonal sampling periods. Overall, the faunal feeding regimes within the lower shoreline were highly similar between the two coastlines and temporal seasonal sampling periods.

Table 6.6. Results of the ANOSIM tests of intertidal faunal species composition a), taxonomic composition b), and feeding regime composition c), between the two coastlines (device; control) and four temporal seasonal sampling periods, within the upper and lower shoreline heights, from 2009 - 2010, Isle of Eday. Tests are based on one-way ANOSIM analyses across each bay site (1-3) between each coastline and temporal seasonal sampling periods (October 2009; April 2010; July 2010; October 2010), separately for each shoreline height. All results were generated on Bray-Curtis similarity resemblance matrices.

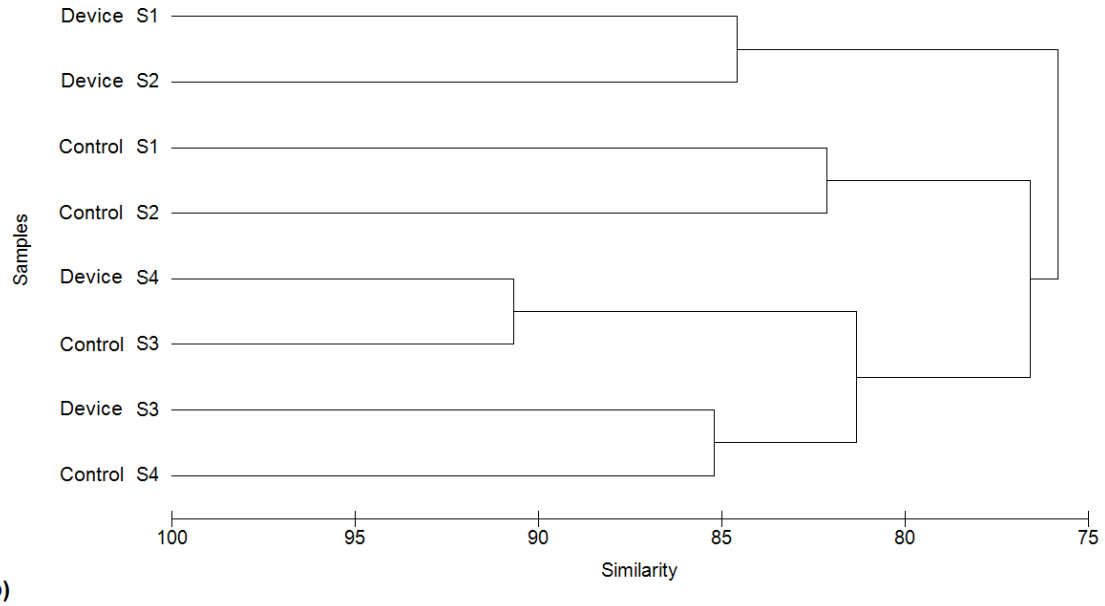
	Global values			
	Upper shoreline		**Lower shoreline**	
	R	***P***	**R**	***P***
a) Species				
Coastline	0.105	0.03	0.05	0.102
Temporal: Device Coastline	0.454	0.005	0.324	0.04
Temporal: Control Coastline	0.435	0.008	0.077	0.25
b) Taxon				
Coastline	0.23	0.007	0.125	0.02
Temporal: Device Coastline	0.231	0.12	0.12	0.18
Temporal: Control Coastline	0.028	0.36	-0.07	0.67
c) Feeding regime				
Coastline	0.313	0.003	0.08	0.06
Temporal: Device Coastline	0.09	0.28	0.136	0.16
Temporal: Control Coastline	0.035	0.31	-0.127	0.76

Table 6.7. Intertidal faunal species composition a), taxonomic composition b) and feeding regime composition c), SIMPER percentage contributions (%) between coastlines in the upper and lower shoreline heights, from 2009 - 2010, Isle of Eday. These are based on the Bray-Curtis average dissimilarity contributions from the one-way SIMPER routines, with bold indicating ≥ 20 %, and average within-group similarities shown.

	Upper shoreline		**Lower shoreline**	
	Device	**Control**	**Device**	**Control**
a) Species				
Cnidaria				
Actinia equina	4.72		**22.02**	19.83
Mollusca				
Gibbula cineraria	0.50		1.32	
Gibbula umbilicalis	0.37			
Littorina littorea	6.16	19.76	3.13	2.28
Littorina obtusata	8.92	**25.77**	0.19	1.57
Littorina saxatilis	1.40	12.34	1.42	3.99
Nucella lapillus	7.56	2.68	**21.59**	15.62
Patella depressa	2.91		11.13	16.71
Patella vulgata	**67.45**	**39.46**	**39.20**	**40**
Average similarity	53.26	35.07	72.02	60.88
b) Taxon				
Cnidaria	5.53		**32.78**	**28.69**
Mollusca	**94.47**	**100**	**67.22**	**71.31**
Average similarity	80.40	81.75	91.63	80.52
c) Feeding regime				
Epifaunal herbivore	**80.24**	**97.99**	**61.40**	**61.76**
Predator	19.76	2.01	**38.60**	**38.24**
Average similarity	82.57	75.65	91.30	82.74



a)



b)

Figure 6.6. Hierarchical cluster analysis dendograms of intertidal faunal species composition across the device and control coastlines and four temporal seasonal sampling periods (1-4) within the upper a) and lower shoreline heights b), from 2009 - 2010, Isle of Eday. Lettering corresponds to the temporal seasonal sampling periods: October 2009 = S1, April 2010 = S2, July 2010 = S3, October 2010 = S4. Cluster analysis is based on the Bray-Curtis similarity resemblance matrix with group average linking.

6.5. Discussion

Macroalgae assemblages

Classical species diversity estimates and exploratory descriptors can widen our knowledge of species assemblage structures within the marine environment, particularly in sites that are under-sampled or under review for new anthropogenic developments. My results first show that both coastlines are dominated by Phaeophyta species. These are large perennial species, which form multi-layered or canopy structures that show continuous algae cover (Rindi and Guiry 2004; Krause-Jensen et al. 2007). Such species provide important ecological functions including refuge, spawning, or feeding opportunities for a variety of intertidal species (Kraufvelin 2007). These results suggest they exhibit morphological forms and that are designed to withstand large, frequent mechanical stress and displaying strong attachment characteristics (Pratt and Johnson 2002; Schiel 2004; Nishihara and Terada 2010). Such physiological traits are ideal within these particular dynamic environments at both shoreline heights; to tolerate the intense hydrodynamic regimes and exposure present.

Throughout this study, my results show increased macroalgae diversity estimates across the device site coastline. These patterns follow other studies, which describe how anthropogenic activities alter ecological patterns within the marine environment (Payne et al 2008; Vaselli et al. 2008; Díez et al. 2009). Identifying such patterns of change is important, as it begins to outline potential ecological interactions with new anthropogenic activities, such as tidal energy developments (Gill, 2005). This information could therefore be extremely valuable for future environmental impact studies and governmental marine energy or environmental policy strategies, including marine developmental licensing processes which may favour anthropogenic activities that actually enhance species diversity (Gill 2005; Inger et al. 2009). In addition, this information, can further aid current knowledge of how anthropogenic activities generally interact with the overall marine environment and also potentially distinguish between them. This is particularly essential, due to the marine environment under severe pressure from a growing range of other anthropogenic activities (fisheries, pollution and shipping) and natural factors (climate change) (Mieszkowska et al. 2006; Eastwood et al. 2007).

My results also show different species composition and morphological forms between the two coastlines, for both shoreline heights. The development site coastline supports species, including *Cladophora rupestris* and *Laminaria digitata*, which show highly opportunistic and

resilient life history characteristics (Steneck and Dethier 1994). These species are often related to highly wave exposed and disturbed environments, either by natural processes, such as storm events, or, anthropogenic activities, such as coastal development (Sousa 1984; Stevens et al. 2002; Wells et al. 2007; Okuda et al. 2010). Interestingly, across the development site coastline, I also recorded the presence of the non-indigenous species, *Codium fragile* subsp. *fragile*. The distribution of this species has been directly linked to anthropogenic activities such as shipping movements, with few records of this species' presence prior to the EMEC development within the Isle of Eday (MNCR 1997; Trowbridge 2001; Aurora 2005: Trowbridge and Farnham 2009). In addition, the development site coastline can also be described as a more degraded intertidal coastal body, compared to the control. This follows coastal body quality ecological descriptors, which use the measured increased proportion of Chlorophyta species and the presence of non-native species, to determine reduced coastal body quality (Wells et al. 2007; Guinda et al. 2008; Díez et al. 2009).

Overall, these observations may represent early indications of ecological response and coastal body status related to tidal energy development activities within intertidal regions. The activities associated with tidal energy development (increased vessel presence; addition of new structures), could therefore promote such species colonisation (species linked to anthropogenic activities) and account for the altered diversity and compositions observed here (Martin et al. 2005). This follows a number of studies which describe altered species diversity and community structure composition from similar marine anthropogenic activities (wave energy; coastal development) or other marine environments (benthic region) (Vaselli et al. 2008; Langhamer 2010; Frid et al. 2012). Macroalgae assemblage composition within the two coastlines may also be subjected to different biological and environmental processes such as water temperature, which were not measured during this study. Therefore, although these results infer some degree of ecological interaction by anthropogenic influence, other underlying processes may be influencing these observations. Determining such causal processes are important to classify and should be included in any future study, particularly in the measurement of new anthropogenic effects and under-sampled environments such as these (Underwood 2000; Okuda et al. 2010). For example, within the control site coastline, I recorded a greater presence of both *A. nodosum* and *F. spiralis*, which are often associated with less wave exposed environments (Menge et al. 2005). Therefore including other environmental variables, such as wave fetch may provide further ecological and

environmental information on this environment (Burrows 2012).The measured temporal scales showed weak signals of macroalgae species diversity and compositional change, with both coastlines comprising of fucoid species, such as *Fucus vesiculosus*. These species are regarded as naturally constrained; due to their limited propagule dispersal, and slow growth rate, which may contribute to these observed temporal patterns (Krause-Jensen et al. 2007). Within the lower shoreline, I also recorded minor increased diversity estimates during the April 2010 sampling period. This pattern can be linked to several seasonal factors, such as increased nutrient input and light availability, and also reduced disturbance regimes i.e. post winter storms or spring tidal cycle conditions (Rindi and Guiry 2004; Schiel 2004; Arévalo et al. 2007; Krause-Jensen et al. 2007). These factors can promote macroalgae species growth, reproduction and competition (Hawkins and Hartnoll 1985; Menge et al. 2005).

In addition, I observed minor increased proportions of understory Rhodophyta division species during October 2010. This follows other studies; which describe their increased presence due to the reduction of other macroalgae species' biomass (such as the Chlorophyta species) during these periods (Schiel 2004; Burrows et al. 2008; Wernberg and Goldberg 2008). Therefore, temporal variation may occur between these measured scales, but represented as minor fluctuations, due to the overlying influence of the abiotic or biotic factors found within this type of dynamic environment. For example, I repeatedly observed filamentous form species such as *Ulva intestinalis* during this study. This species is a highly opportunistic species, known to tolerate disturbance regimes (natural and anthropogenic) and exhibit fast growth rates (Steneck and Dethier 1994; Orfanidis et al. 2001; Menge et al. 2005). This species may show small, yet important temporal responses between the measured scales used here. Therefore future studies could benefit from focusing on specific species, to identify key temporal variation responses within the intertidal environment for both ecological baseline and anthropogenic impact assessments. Furthermore, important ecological temporal variation may also occur at other temporal scales (bi-weekly, or annual periods), which were not included within this study and should therefore be incorporated for future assessments.

Faunal assemblages

The faunal assemblages found within this study further relate to environments that are highly exposed and frequently undergo high levels of disturbance, from strong hydrodynamic

regimes conditions. The type of faunal species I recorded show life history strategies related to increased attachment properties and feeding strategies, which enable them to prevail within these environments (Warburton 1976; Schiel 2004; Shields et al. 2011). Such extreme environments are also commonly represented by few faunal species, which is shown throughout my study; in terms of comparable diversity estimates and composition between the two coastlines (Burrows et al. 2008; Davenport et al. 2011). My results however, also outlined that the development site coastline showed reduced diversity evenness within both shoreline heights, and, compositional differences within the upper shoreline. This was due to the dominance and presence of species such as *Patella vulgata* and *Actinia equina*. *A. equina* is regarded as a filter feeder, but is also known as an generalist, with scavenging and predatory traits, that are suitable to rapid natural (such as wave exposure) and anthropogenic changes (such as coastal development) (Dalton 2008; Davenport et al. 2011). Overall, these subtle diversity and compositional differences may give a true reflection of the dissimilar underlying environmental parameters (such as wave exposure), biological interactions or the added of anthropogenic use between the two coastlines at the faunal scale (Okuda et al. 2010; Burrows 2012).

The recorded faunal assemblages also showed minor diversity and compositional changes across the study's measured temporal scales, within both shoreline heights. This could reflect the similarity of the underlying abiotic and biotic processes occurring within these regions; measured across these temporal scales, and also the type of dominant faunal species present (Alfaro and Carpenter 1999). I also observed reduced diversity evenness estimates during April and October 2010 and species compositional differences across the sampling periods within the upper shorelines. These patterns can be related to the temporary presence of *Littorina* species (see Appendix 5 Table A.5.4) I recorded, which can also be associated to a range of biotic and abiotic factors, which are known to alter over time (Menge et al. 2005). During periods of increased hydrodynamic events (winter storms or spring tides), *Littorina* species are driven into other areas for refuge or alternative food sources, whilst during summer periods, their abundance and distribution can increase for feeding or reproduction opportunities (Alfaro and Carpenter 1999; Guera-García et al. 2011). Their temporary presence may also be related to the type of macroalgae species present and their affiliated morphological form present for feeding opportunities. A number of studies have recorded that these mollusc species show particular feeding preference for foliose type macroalgae, such as *Ulva* spp, which I recorded within both shoreline heights, during the April and July

sampling periods. This type of macroalgae functional form is more favourable to these herbivores; due to the high surface area to volume ratio, increased nitrate saturation levels and also comprise of reduced levels of secondary metabolites, which can deter grazing (Krause-Jensen et al. 2007; Guerry et al. 2009; Cacabelos et al. 2010). Overall, this suggests that despite the intense hydrodynamic regimes and potential anthropogenic influences, complex biotic relationships occur between the macroalgae and faunal communities within this intertidal environment.

Conclusions and future study considerations

Overall, my results begin to outline that the intertidal macroalgae and faunal communities within the coastlines of the Isle of Eday are part of a complex, highly dynamic disturbance system. This is shown by their competitive opportunistic growth, adaptive body plans and feeding tendencies (Menge et al. 2005). Physiological tolerance and quick recovery life strategies are particularly important for such species' living within this type of hydrodynamic environment (Steneck and Dethier 1994; Schiel 2004).

In terms of their ecological interactions with the anthropogenic activities associated to the tidal energy test site, these intertidal communities indicate varied or minor species diversity and compositional responses. Macroalgae assemblages showed stronger ecological responses to these anthropogenic activities at both shoreline heights. This may be a result of their sessile and morphological characteristics, as they are more exposed to such anthropogenic influences. Faunal assemblages showed possible subtle ecological responses between the two coastlines, particularly within the upper shoreline level. Overall, this highlights the importance of selecting appropriate survey objectives, ecological units, and scales and methods (statistical and field-based), when designing monitoring strategies for new anthropogenic activities (Carballo and Naranjo 2002; Shields et al. 2009). Therefore, the combined use of both macroalgae and faunal assemblages and measureable scales (diversity, composition and functional form) to assess anthropogenic activities upon the intertidal regions are advantageous for future environmental impact assessments; to first outline the ecological interactions at the species scale and then across the intertidal region as a whole. In addition, these particular coastlines could also be used as future baseline sites for ecological and anthropogenic comparison studies. Selecting comparable environments is important,

especially due the number of future tidal energy development schemes proposed within this offshore area and the Orkney Isles (Gill 2005; Norris and Droniou 2007).

Further on-going seasonal macroalgae and faunal community assessments using these methods and coastline sites are first recommended, to increase baseline information and the potential ecological interactions with new tidal energy developments (Shields et al. 2011). In particular, extending this study at the spatial scale (Isle of Eday and adjacent islands) and the temporal scale (seasonal, annual) may further reveal important biotic, abiotic and anthropogenic interactions that may have been missed and increase sampling efforts. The statistical methods used, gave valuable univariate and multivariate results for this study. The SIMPER routines however, presented a number of hidden patterns, which may potentially be important, ecologically, for example: the minor compositional differences between the macroalgae taxonomic divisions. Therefore displaying other ecological information, such as total macroalgae percentage cover or faunal abundance results may be more useful. In addition other complementary methods are also recommended for such future assessments. At the macroalgae level this could include using coastline habitat quality indices (such as the CFR index), or focussing at specific indicator species such as the invasive *Codium fragile* subsp. *fragile* and key-stone species such as *Fucus vesiculosus*; in terms of distribution and presence related to tidal energy development activities (Wells et al. 2007; Guinda et al. 2008; Trowbridge and Farnham 2009). At the faunal scale, I suggest a more fine-scale approach; in terms of the taxonomic and functional form compositional assessments. The use of higher taxonomic resolution levels such as the Phylum level, to measure natural or anthropogenic change has been used in several marine, estuarine and sub-tidal environment studies (Bates et al. 2007; Wells et al. 2007). Where, however, environments have few species, such as these coastlines, this coarser taxonomic assessment may be inappropriate. In this context, I propose that the intertidal faunal community structures within these coastlines should be either assessed at the species taxonomic level or where rapid or basic environmental impact assessments are required, at the taxonomic order level. In terms of faunal functional feeding regimes, these categories may also be too generalised and may miss important, subtle responses to biotic, environmental or anthropogenic influences. For example, different types of epifaunal herbivores occur based on intrinsic properties such as species bite penetration depth, or preference to specific algae species (Steneck and Dethier 1994). Therefore a more defined feeding regime category could be developed, with these factors taken into consideration, which may also reveal further interesting interactions with the macroalgae

assemblages within these coastlines. I also recommend that future studies also focus on specific biotic relationships between the macroalgae and faunal assemblages. In particular, such studies should consider the relationships between macroalgae assemblages and key faunal species such as the macroalgae *Ulva intestinalis*, the herbivore *P. vulgata*, and the predator *N. lapillus*. This could also include barnacle species, such as *Semibalanus balanoides*, which I also observed within the lower shoreline during this study (total percentage coverage of *Semibalanus balanoides* across the two coastlines < 3%). These colonial species are suspension feeders, which show structural complexity, associated with strong tidal flows and herbivore/predator intensity (Gili and Coma 1998; Schiel 2004). Including this species within these biotic interaction assessments, may outline more direct, accurate relationships with the environmental conditions found within these extreme environments.

Finally, I recommend a comparative review of the type and frequency of the anthropogenic activities that are conducted within the development site. Combined with the ecological studies suggested above, this could specifically indicate more direct and indirect ecological responses to anthropogenic activities associated with tidal energy development. This could include assessing species presence and distribution with vessel activities, species compositional change from device deployments or altered hydrodynamics from device energy generation operations. Overall, a more definitive review of the range of anthropogenic activities with measured ecological response within the intertidal system could aid future environmental impact assessments to tidal energy development.

In general, the use of intertidal regions to assess such ecological impacts from offshore anthropogenic activities may be extremely valuable in the long term. The areas selected for offshore marine tidal energy developments are often extremely difficult to sample in effectively, due to the increased hydrodynamic regimes which reduce offshore ecological sampling efforts (Shields et al. 2011). As such, this region could represent an important surrogate for ecological study of the development of tidal energy within the marine environment. Selecting regions that can detect such interactions with new anthropogenic activities, in environments that are described as hostile, and with few records is a notable challenge. Such studies are important, especially due to the future large-scale expansion of such renewable energy developments within the marine environment.

Chapter 7. Conclusion

7.1. Summary of results

This thesis aimed to explore the benthic and intertidal ecology of the marine tidal race environment, and examine the environmental impacts from tidal energy developments, using field-based studies.

Within the Alderney tidal race (Chapter 2), I found that the benthic regions are topographically complex, due to the variety of physical seafloor components present. This fine scale approach outlined comparable physical and ecological components between the survey locations, which altered temporally. I related this to the extreme and variable nature of the hydrodynamic regimes influencing the physical and ecological components present (Hiscock 1983; Denny 2006). These results also outlined the potential role of other marine processes that could occur within this environment type, including disturbance, succession and recruitment scenarios (Balata et al. 2006; Callaway et al. 2009; Storlazzi et al. 2011).

The intertidal regions of this marine tidal race environment showed spatial and temporal heterogeneity in macroalgal and faunal species biodiversity, and community composition across the shoreline (Chapter 3). Here, I also associated this ecological variability with the complex interactions between environmental and biotic factors that are commonly found within the intertidal environment (Underwood 2000; Menge et al. 2005). My species colonisation experiment further determined the mechanism of intertidal ecological succession, and also the overall types of opportunistic, resilient or mobile life history traits found within this type of environment (Sousa 1979, 1984; Dean and Connell 1987a, b; Chapman 2002).

Overall, these two chapters provide new insights into the ecology of the marine tidal race environment. In particular, it aids the ecological characterisation of the benthic and intertidal regions primarily of these two marine tidal race environments, in-terms of species diversity, community composition and structure. This also builds on from past studies that describe how species presence, assemblage composition and life history traits, are strongly related to water movements and other underlying environmental and biotic processes (Hiscock 1983; Connor et al. 2004; Menge et al. 2005; Burrows et al. 2008; Shields et al. 2011). My research

furthers these, by outlining how the ecology within these types of environment can vary over space and time.

My results also revealed a number of environmental impacts associated with the tidal energy development activities within the EMEC tidal energy test development site, Orkney. Here, I found that a single tidal energy device structure can act as a temporary aggregation site for shoals of *Pollachius pollachius*, potentially for feeding or reproduction (Chapter 4; Selleslagh and Amara 2008; Seppälä et al. 2008). Interestingly, this study also revealed the relationship between species presence and the hydrodynamic regimes, with increased tidal velocity leading to reduced fish abundance.

In Chapter 5, my benthic ecological study showed comparable results with other past renewable energy environmental impact research. These studies and my own, suggest the potential for these devices to act as localised artificial reef structures, which can increase benthic biodiversity estimates and influence species composition (Chapter 5; Wilhelmsson and Malm 2008; Langhamer 2010; Langhamer et al. 2010). This study identified that species composition altered seasonally, under the potential influence of a range of abiotic and biotic processes. Furthermore, I found that the marine habitat types present within this region were strongly associated to the intense hydrodynamics conditions and proximity to the coastline overall.

The intertidal regions of the development site (Chapter 6) also showed some ecological differences from the control site. This was identified by the added presence of ephemeral and invasive species within the development site, which are associated with anthropogenic environmental impacts, such as coastal development activities (Trowbridge 2001; Wells et al. 2007). The overall macroalgae and faunal species composition and life history traits were however strongly related to the dynamic environment, which altered over time and space (Okuda et al. 2010).

Taken together, Chapters 4, 5, and 6 reveal that the level of anthropogenic impact from current tidal energy development activities within this environment is scale dependent (both in space and time), and related the type of region, or ecological aspects (for example, species composition) chosen for study. My results suggest that these measured ecological aspects within this marine tidal race environment are still primarily influenced by the extreme hydrodynamic conditions and other environmental and biotic processes, which require further study.

All chapters combined present the variable patterns of ecology within marine tidal race environments. This research provides quantitative information which builds upon the current knowledge of this uncommon, extreme marine environment. Fundamentally, this study outlines the overall complex relationships between ecology and the changing nature and influence of both natural and new anthropogenic processes.

It must be noted, however that these overall interpretations may also be a result of a number of caveats and limitations I found during this study.

7.2. Study caveats

The limitations of this study are primarily affiliated with the extreme hydrodynamic regimes, weather conditions and topographical complexity of sampling within marine tidal race environments. Despite adaptions of the field-based sampling techniques used, and selecting appropriate levels of spatial and temporal scales to investigate these environments (i.e. during neap tidal cycles (Chapters 2 and 5) or summer periods (Chapter 4)), the intense environmental conditions still reduced my sampling efforts. These factors hindered spatial and temporal scale sampling, observation of ecological behaviour, and also led to the loss and breakage of sampling gear. This limitation may therefore affect the overall ecological interpretation and characterisation of this environment type (characterisation: species diversity, composition and community structure), and in turn, the ecological consequences of tidal energy development. Therefore modifications of my study methods or alternative sampling designs may be required to effectively quantify this challenging environment in the future.

The spatial and temporal scales that I originally selected for study may also limit the ecological interpretation of this environment. The spatial scales used here were small in comparison to the overall size of the two studied marine tidal race environments. Further ecological responses may occur over longer or different temporal scales, which could not be assessed within the timeframe of this research (Wernberg and Goldberg 2008). In addition, the assessment of impacts should explore responses to different spatial and temporal scales of the development itself, including the number of devices deployed, overall infrastructure deployment, maintenance and the tidal energy device life cycle (Shields et al. 2009).

My observations are also restricted by the sampling methods used. I employed robust observational techniques that proved to cope well with the intense hydrodynamic regimes: underwater video camera surveys, interpretation of sampled still images from video and generalised habitat descriptions. These gave quantitative ecological information, but preclude the identification of small or taxonomically difficult organisms, such as cryptic bryozoan and hydroid species. The relatively narrow viewing width, along with time limits on the number of surveys and the time required for image processing, also precluded the broader spatial or temporal replication (Brown et al. 2011; Ierodiaconou et al. 2011). I also did not record environmental variables such as water temperature and salinity, which can provide greater accuracy in defining the ecological patterns within this environment but would also require more temporal replication to capture variation in these variables (Okuda et al. 2010; Rodrigues et al. 2011).

Although I studied a range of ecological components, across differing regions, there may be other ecological components that would help further describe this habitat and the environmental impacts from tidal energy development. This could include: other marine species (i.e. seabirds); species functional life history traits (intertidal herbivore feeding mechanisms); biotic relationships (intertidal barnacle mosaics with algal and faunal species); or the interactions between environmental and biotic processes (hydrodynamic regimes and larval dispersal), which should all be considered for future study (Gili and Coma 1998; Schiel 2004; Martin et al. 2005: Inger et al. 2009).

The fundamental caveat of this research is, however, the problem of sampling replication for the overall ecological interpretation of this environment type. The research strategy, sampling method designs and derived results gave new and valuable insights into these two marine tidal race environments, in terms of species diversity, composition and potential influences with current tidal energy device developments. This study, however, only investigated two marine tidal race environments and derived the environmental impacts observations, surrounding one tidal energy device and development site. Whilst this information is particularly useful for these two specific sites, I would urge caution to use this information to describe the overall environment type (i.e. full, in-depth community composition information of this environment type) and the total potential scale of environmental impacts from tidal energy development. This lack of replication stems from the natural uncommonness of this environment, limited past sampling strategies and relative novelty of the technology: few tidal devices are in active use and new projects, such as that planned for the Alderney Race,

are often subject to considerable delay with inevitable consequences for planned sampling. This is of particular importance as marine tidal race environments are generally under-sampled, with tidal energy technology and subsequent development plans still considerably new (Gill 2005; Inger et al. 2009; Shields et al. 2011). As such, the ecology, and the patterns observed (spatial and temporal) within other marine tidal race environments may vary and therefore also the affiliated environmental impacts from tidal energy development. In addition, caution should be directed at the application of my research for future large-scale tidal energy development impact analyses. How such large-scale developments may impact the present marine environment is, as yet unknown, with the debate only just beginning (Kirby 2010; Neill et al. 2012). To overcome this lack of replication is an important future challenge, particularly as these environments are uncommon in nature, difficult to sample in and that the locations for tidal energy development are also site specific. For this I outline a number of recommendations for the direction of future study.

7.3. Direction for future study

The direction for future study should aim to further advance and expand our understanding of the ecological aspects and characterisation of marine tidal race environments. In addition, the increasing growth of tidal energy development within this environment also warrants further environmental impact investigation (Gill 2005; Langhamer 2010; Shields et al. 2011). Although the scope for future ecological research is extremely wide, I recommend that future studies first expand from my research here, and then extend to develop a wider ecological perspective of this environment type using appropriate, adaptable methods of study. For this I outline a number of recommendations that could be used for future study.

First, I recommend the continuation of my field-based studies, within the two selected marine tidal race environments. This would improve the replication of my work, and increase the ecological knowledge of these environments in the longer term. I suggest that the spatial or temporal scales I originally selected however should also be expanded. I would recommend extending the sampling units within and beyond the benthic and intertidal regions of this environment. This could include increasing benthic video /potting lines, intertidal transect lines/ bay sites, and marine regions type (i.e. water depths: 5 – 10 metres). I would also recommend full tidal cycle assessments within the benthic zones (Chapters 2 and 5), bi-

weekly sampling within the intertidal lower shoreline (Chapters 3 and 6), or annual sampling of fish species' behavioural response (Chapter 4) in order to capture temporal variation.

These future studies would also benefit from the inclusion of complementary sampling methods and research investigations. This could comprise of scuba-dive methods or species tagging methods, to investigate tidal energy device structures acting as artificial reefs (Chapters 4 and 5) or, studies could include species indicator indices, to assess ecosystem integrity (Chapter 6) (Wells et al. 2007; Antoniadou et al. 2011; Brown et al. 2011). Furthermore, my original method designs were primarily descriptive and observational based. Although these methods were useful to first define the ecology of this environment, the next obvious step would be to implement experimental or manipulation ecological investigations (Underwood 2000; Underwood et al. 2000). These could test key ecological theories such as succession, the role of disturbance and life history strategies (Chapters 2 and 3; Sousa 1984; Tilman 1990; Steneck and Dethier 1994; Nishihara and Terada 2010).

Throughout this study, I linked ecological patterns to key environmental and biotic processes that are known to occur within the marine environment. Therefore, I also recommend that future studies should further incorporate key environmental and biotic variables that occur within this environment. This could include; velocity rate (both vertical and horizontal measurements), wave fetch, bed sheer stress, water temperature, ecological trophic interactions or larval dispersion factors. Combining these factors and complementary methods with the ecological components I selected during this study, would enable a more in-depth review of the influencing and indeed changing roles between the environmental, biotic and anthropogenic processes within this environment (Okuda et al. 2010; Ierodiaconou et al. 2011; Rodrigues et al. 2011). This would also be extremely beneficial for marine habitat mapping classification recording schemes such as JNCC or EUNIS. These studies could expand existing information, by improving the basic physical hydrodynamic descriptions and ecological knowledge base, which may also create new classification levels for this environment type (Connor et al. 2004).

I strongly recommend that the direction of future study should involve both field-based methods and computerised modelling techniques (Chapter 2). Integrating a range of different yet appropriate methods could increase our overall understanding of this environment type, but more importantly attempt to deal with the problems associated with the present hydrodynamic conditions and replication efforts. For example, the relationship between

species' spatial or temporal distributions (such as *T. indivisa* or *P. pollachius*) with velocity rates could be further investigated, using both field based studies and computerised models, such as 3D POLCOMS or TELEMAC (Chapter 2; Neill et al. 2012). This could involve studies which examine temporal interactions between scour and benthic assemblage composition, across different tidal cycles (neap and spring scenarios) and temporal scales (monthly, seasonal and annual). In addition, a comparative study of European marine tidal race environments using advanced methods such as bathymetry mapping techniques, combined with past and current ecological (both anecdotal and quantitative information) research could enable a definitive ecological assessment of these environments (Burrows et al. 2012). This may then aid other studies which aim to quantify the ecological influences from other hydrodynamic regime conditions (such as intense wave exposure environments) (Burrows et al. 2012). For example, this approach could therefore include a comparative review of a range of water movement scenarios (extreme tidal flow vs. extreme wave exposure) upon the present ecological communities, which may further define the ecology of these different extreme environments overall.

Furthermore, this approach could provide a more accurate method to determine the environmental impacts from current and future tidal energy developments. For example, this approach could be used to investigate and define ecological impacts scenarios across different tidal energy device life cycles (installation, operational and decommissioning stages) and development scales (1-10; 10 – 50; 50 + device arrays deployed within in one site) (Bailly du Bois et al. 2012; Neill et al. 2012). This would be extremely important, given the tidal energy industry at an early pre-commercial development stage, with future large scale development plans stage still in development (Gill 2005; Shields et al. 2011; Bahaj 2011). In addition, this approach could outline the true impact from such developments, in terms of negative (habitat loss, collision risk) or positive (diversity enhancement, protected sites) impacts upon the marine environment, with such impact knowledge still unknown, primarily descriptive, or strongly conflicted (Gill 2005; Inger et al. 2009). Therefore, using such a wider-scaled approach could enhance replication efforts within these uncommon environments and in turn, further define them and the anthropogenic impacts that they may face in the future.

Overall, my results highlight the use of these two marine tidal race environments as key reference sites of this environment type for future ecological investigations. I primarily recommend, however the development of a more detailed, yet, wide-scaled ecological perspective of this environment type. This should entail ecological comparisons with other

marine tidal race environments and tidal energy development locations, such as the Pentland Firth (Scotland), Ramsey Sound (Wales), Isle of Wight, Big Russell (Guernsey) and Paimpol (France) to enhance current knowledge (i.e. in-depth quantitative descriptions community information). To undertake a review such as this, my final recommendation is to propose further and more in-depth research collaborations and shared resources (information, field-based surveys etc) between key academic, scientific and industry bodies. Currently a number research groups exist, which are either geographically bound (Orkney Isles or South-West of England) or renewable energy device specific (tidal, wave or wind energy device types). A more wider-scaled collaboration between groups such as EMEC, IFREMER, PRIMARE and MREDS would allow more in-depth ecological research and shared knowledge base into these particular environments. Furthermore, this research collaboration should extend to the tidal energy industry, including device manufacturers, developers, marine spatial planners and environmental impact assessors (Scott 2007). This could instigate more appropriate and comparative environmental impact research assessments i.e. between different tidal energy devices, test or commercial development sites or other offshore renewable energy devices (i.e. wave energy devices). An additional wide-scale ecological meta-analysis of these offshore anthropogenic activities would be inherently useful, to understand the varied degrees of environmental impacts within the marine environment. This could enable exciting new ecological and environmental research possibilities, whilst solving problems associated with sampling this extreme environment in terms of; sampling replication, limited resources or knowledge and surveying costs. Such a collaborative approach may be able to fill in the missing gaps on the ecology and accurately identify anthropogenic impacts within this environment, which are currently of considerable ecological concern (Inger et al. 2009). This is extremely important given their potential wide-scale development across the marine environment in the future. Currently, the global energy sector favours this new source of energy, and commonly regards these technologies as ‘green’ or ‘environmentally benign’, compared to other sources within the marine environment (such as oil and gas exploration) (Gill 2005). This is due to these technologies seen as alternatives to dwindling fossil fuel energy sources, providing long-term sustainable energy to growing future energy demands, whilst reducing carbon dioxide (CO^2) emissions (Asif and Muneer 2007). At present, however, the marine environment is under significant pressure from other anthropogenic factors, such as fisheries, pollution, shipping and resource extraction (Eastwood et al. 2007). Therefore understanding the true, added impact of this new type of anthropogenic activity upon the marine environment is paramount. Ultimately a combined multidisciplinary

approach between marine ecology, oceanography and industry, would increase the current lack of ecological knowledge of marine tidal race environments and the associated environmental impacts from the growing development of offshore tidal energy, in the long term.

7.4. Concluding remarks

The ecology within the benthic and intertidal regions of these marine tidal race environments show variable patterns over space and time. Consequently the degree of environmental impacts associated with current tidal energy development activities also varies. Overall, future studies are vital; to further advance ecological knowledge of this uncommon marine environment and the interactions with such new anthropogenic activities. Future studies should combine a range of applicable research methods, whilst not forgetting the fundamentals of marine ecology, the challenging nature of this environment type or the future expansion of renewable energy development in the marine environment.

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Appendix 1. Supporting information for chapters 2, 3, 4, 5 and 6.

Table A.1. Benthic and intertidal species presence and functional form recorded during the Alderney (Channel Islands) and Isle of Eday (Orkney Isles) studies, 2009 - 2011.

Species	Case-study Location		Functional Form
	Alderney	Isle of Eday	
Marine Algae			
Division: Chlorophyta			
Blidingia minima (Nägeli ex Kützing) Kylin, 1947	X	X	FA
Chaetomorpha linum (O. F. Müller) Kützing, 1845	X		FA
Cladophora rupestris (Linnaeus) Kützing, 1843	X	X	FA
Codium fragile subsp *fragile* (Suringar) Hariot, 1889	X	X	FA
Ulva intestinalis Linnaeus, 1753	X	X	FA
Ulva lactuca Linnaeus, 1753	X	X	FS
Ulva linza Linnaeus, 1753	X		FA
Division: Phaeophyta*			
Ascophyllum nodosum (Linnaeus) Le Jolis, 1863	X	X	LM
Bifurcaria bifurcata R. Ross, 1958	X		LM
Colpomenia peregrina Sauvageau, 1927	X		FA
Cystoseira tamariscifolia (Hudson) Papenfuss, 1950	X		LM
Dictyota dichotoma (Hudson) J. V. Lamouroux, 1809	X		FC
Ectocarpus siliculosus (Dillwyn) Lyngbye, 1819	X		FA
Fucus serratus Linnaeus, 1753	X	X	LM
Fucus spiralis Linnaeus, 1753	X	X	LM
Fucus vesiculosus Linnaeus, 1753	X	X	LM
Halidrys siliquosa (Linnaeus) Lyngbye, 1819	X		LM
Himanthalia elongata (Linnaeus) S.F. Gray, 1821	X	X	LM
Laminaria digitata (Hudson) J.V. Lamouroux, 1813		X	LM
Laminaria hyperborea (Gunnerus) Foslie, 1884	X		LM
Pelvetia canaliculata (Linnaeus) Decaisne & Thuret, 1845	X	X	LM
Sargassum muticum (Yendo) Fensholt, 1955	X		LM
Division: Rhodophyta			
Ahnfeltia plicata (Hudson) E.M. Fries, 1836	X		CM
Asparagopsis armata (Harvey), 1855	X		FP
Catenella caespitosa (Withering) L.M. Irvine, 1976	X		CM
Ceramium virgatum Roth, 1797	X	X	FP
Chondrus crispus Stackhouse, 1797	X	X	CM
Corallina elongata J. Ellis & Solander, 1786	X		AC
Corallina officinalis Linnaeus, 1758	X	X	AC
Dilsea carnosa (Schmidel) Kuntze, 1898	X		FS
Drachiella spectabilis J. Ernst & Feldmann, 1957	X		FS
Dumontia contorta (S. G. Gmelin) Ruprecht, 1850	X		CM
Furcellaria lumbricalis (Hudson) J. V. Lamouroux, 1813	X		CM
Gelidium spinosum (S. G. Gmelin) P. C. Silva, 1996	X		CM
Gelidium pusillum (Stackhouse) Le Jolis, 1863	X		CM
Gracilaria gracilis (Stackhouse) M. Steentoft, L. M. Irvine & W.F. Farnham, 1995	X		CM
Osmundea pinnatifida (Hudson) Stackhouse, 1809	X	X	CM
Laurencia obtusa (Hudson) J. V. Lamouroux, 1813	X		CM

Species	Case-study Location		Functional Form
	Alderney	Isle of Eday	
Lithophyllum species Philippi, 1837	X	X	CA
Lomentaria articulata (Hudson) Lyngbye, 1819	X		CM
Nemalion helminthoides (Velley) Batters, 1902	X		CM
Division: Rhodophyta			
Mastocarpus stellatus (Stackhouse) Guiry, 1984	X		CM
Palmaria palmata (Linnaeus) Kuntze, 1891	X		FS
Plumaria plumosa (Hudson) Kuntze, 1891	X		FP
Porphyra umbilicalis Kützing, 1843	X	X	FS
Rhodomela confervoides (Hudson) P. C. Silva, 1952	X		FP
Rhodothamniella floridula (Dillwyn) Feldmann, 1978	X	X	FA
Vertebrata lanosa (Linnaeus) T. A. Christensen, 1967	X	X	FP
Division: Tracheophyta			
Zostera (Zostera) marina Linnaeus, 1753	X		A
Marine Fauna			
Phylum: Annelida			
Eulalia viridis Linnaeus, 1767	X		O
Janua (Dexiospira) pagenstecheri de Quatrefages, 1865	X		SF
Spirorbis (Spirorbis) spirorbis Linnaeus, 1758	X	X	SF
Phylum: Arthropoda			
Sub-Phylum: Crustacea			
Balanus crenatus Bruguiére, 1789	X		SF
Cancer pagurus Linnaeus, 1758	X	X	O
Carcinus maenas Linnaeus, 1758	X	X	O
Chthamalus montagui Southward, 1976	X		SF
Hyas coarctatus Leach, 1816		X	O
Necora puber Linnaeus, 1767		X	O
Pagurus bernhardus Linnaeus, 1758		X	O
Palaemon elegans Rathke, 1837	X		O
Portumnus latipes Pennant, 1777	X		O
Semibalanus balanoides Linnaeus, 1758		X	SF
Sub-Phylum: Hexapoda			
Anurida maritima Guérin-Méneville, 1836	X		O
Phylum: Bryozoa			
Electra pilosa Linnaeus, 1767	X		SF
Flustra foliacea Linnaeus, 1758	X		SF
Phylum: Chordata			
Conger conger Linnaeus, 1758		X	P
Pleuronectes platessa Linnaeus, 1758		X	P
Pollachius pollachius Linnaeus, 1758		X	P
Scorpaena porcus Linnaeus, 1758		X	P
Phylum: Cnidaria			
Actinia equina Linnaeus, 1758	X	X	P
Actinia fragacea Tugwell, 1856	X		P
Actinothoe sphyrodeta Gosse, 1858	X		O
Alcyonium digitatum Linnaeus, 1758	X		P
Anemonia viridis Forskål, 1775	X		P
Corynactis viridis Allman, 1846	X		O
Hydroid species	X		SF
Metridium senile Linnaeus, 1761	X		SF

Species	Case-study Location		Functional Form
	Alderney	Isle of Eday	
Tubularia indivisa Linnaeus, 1758	X		SF
Urticina felina Linnaeus, 1761	X		O
Phylum: Echinodermata			
Asterina gibbosa Pennant, 1777	X	X	O
Asterias rubens Linnaeus, 1758	X	X	P
Crossaster papposus Linnaeus, 1767		X	P
Echinus esculentus Linnaeus, 1758		X	P
Phylum: Mollusca			
Angulus tenuis da Costa, 1778	X		SF
Buccinum undatum Linnaeus, 1758		X	P
Calliostoma zizyphinum Linnaeus, 1758	X		O
Gibbula cineraria Linnaeus, 1758	X	X	EH
Gibbula pennanti Philippi, 1846	X		EH
Gibbula umbilicalis da Costa, 1778	X	X	EH
Haliotis tuberculata Linnaeus, 1758	X		EH
Lacuna pallidula da Costa, 1778	X		EH
Littorina compressa Jeffreys, 1865	X		EH
Littorina fabalis Turton, 1825	X		EH
Littorina littorea Linnaeus, 1758	X	X	EH
Littorina obtusata Linnaeus, 1758	X	X	EH
Littorina saxatilis Olivi, 1792	X	X	EH
Melarhaphe neritoides Linnaeus 1758	X		EH
Mytilus edulis Linnaeus, 1758		X	SF
Nassarius reticulatus Linnaeus, 1758	X		P
Nucella lapillus Linnaeus, 1758	X	X	P
Osilinus lineatus da Costa, 1778	X		EH
Patella depressa Pennant, 1777	X	X	EH
Patella pellucida Linnaeus, 1758	X		EH
Patella vulgata Linnaeus, 1758	X	X	EH
Philine aperta Linnaeus, 1767	X		P
Phylum: Porifera			
Amphilectus fucorum Esper, 1794	X		SF
Clathria (Microciona) atrasanguinea Bowerbank, 1862	X		SF
Cliona celata Grant, 1826	X		SF
Halichondria (Halichondria) panacea Pallas, 1766	X	X	SF
Hymeniacidon perlevis Montagu, 1818	X		SF
Myxilla (Myxilla) incrustans Johnson, 1842	X		SF
Pachymatisma johnstonia Bowerbank in Johnston, 1842	X		SF
Tethya citrina Sarà & Melone, 1965	X		SF

Notes: X denotes species presence. This includes all benthic and intertidal macroalgae and species recorded for all study chapters. * represents Ochrophyta phylum species but uses the Phaeophyta phylum descriptions for these studies, for ease of interpretation. Macroalgae functional form is based on the morphological form type: FA = filamentous algae; FP = filamentous (2); FS = foliose single; FC = foliose corticated; CM = corticated macrophyte; LM = leathery macrophyte; AC = articulated calcareous; CA = crustose algae; A = angiosperm. Benthic and intertidal species morphological form is based on the feeding regime type: EH = epifaunal herbivore; O = omnivore; P = predator; SF = suspension feeder.

Appendix 2. Supporting information for chapter 2.

Table A.2.1. Towed camera video system specifications.

Specifications	Details
Vessel survey configuration Water depth measurement	On-board vessel water depth echo-sounder
Geographical Positioning System (GPS)	x 1 SeaTrack™ GPS video overlay (PAL-B European standard) system x 1 Garmin GPS unit (on-board vessel GPS were also used for all surveys)
Data monitor	x 1 7" monitor x 1 10" monitor
Video recorder	x 1 SONY multi-function DVD recorder (SeaViewer Inc, custom built version), PAL-B standard (video-out European requirements) with additional built in monitor (7"). Recorder: DVD, internal and external SD memory card recording. x 1 DVR recorder – 8 GB SD card (PAL-B standard version)
Towed camera video system Camera type (custom built)	Seaviewer Inc 950 Sea-Drop camera
Camera	4.3 mm
Pixels	768 x 464 (based on NTSC information not PAL-B)
Video and photograph output type	Video: AVI. Files; Photograph: bmp.files
Photographic field of view dimensions	1.25 m^2 (auto-focus)
Colour	True day colour, with additional black/white option
Camera lighting	Internal (with additional lighting (LED) custom built)
Camera dimensions	3" diameter x 7-1/4" length
Dry weight	1.24 kilograms
Wet weight	0.49 kilograms negatively buoyant
Umbilical power cable type	SeaSerpent™
Umbilical power cable properties	Polyethylene insulated, 1/4" diameter
Umbilical cable length	70 meters
Safety line properties	Marlow multi-plait rope, 8mm
Safety line length	80 meters
Towed Frame Frame dimensions	0.89 m (length) x 0.4/ 0.43 m (width) x 0.29 m (height)

Frame properties and information	The towed frame is designed on a paravane design. The frame is built out of plastic, which was moulded and adapted from an original sled design. The wings, safety line positions and the lead weight were added to the frame to stabilise the camera system underwater (following Service and Golding 2001; MESH 2008). The design and materials used were selected to protect the camera and increase ease of deployment and type of benthic surveying activities.
Lead weight	1.36 kilograms
Additional information	Additional water depth measurements were marked onto the umbilical and safety lines (in 5 m increments) which included a line out on the frame.
Power requirements Operating voltage for all survey requirements	DC 12 V (nominal) (PAL-B standard)
Power source: vessel based	Water depth measurements: water depth echo-sounder x 1 GPS unit (external on-board vessel unit)
Power source: external battery based:	x 1 Garmin GPS unit x 2 Data monitor x 2 Video recorders x 1 Camera video system

Figure. A.2.1. Schematic diagram of the towed camera system (not to scale, and adapted from Service and Golding 2001).

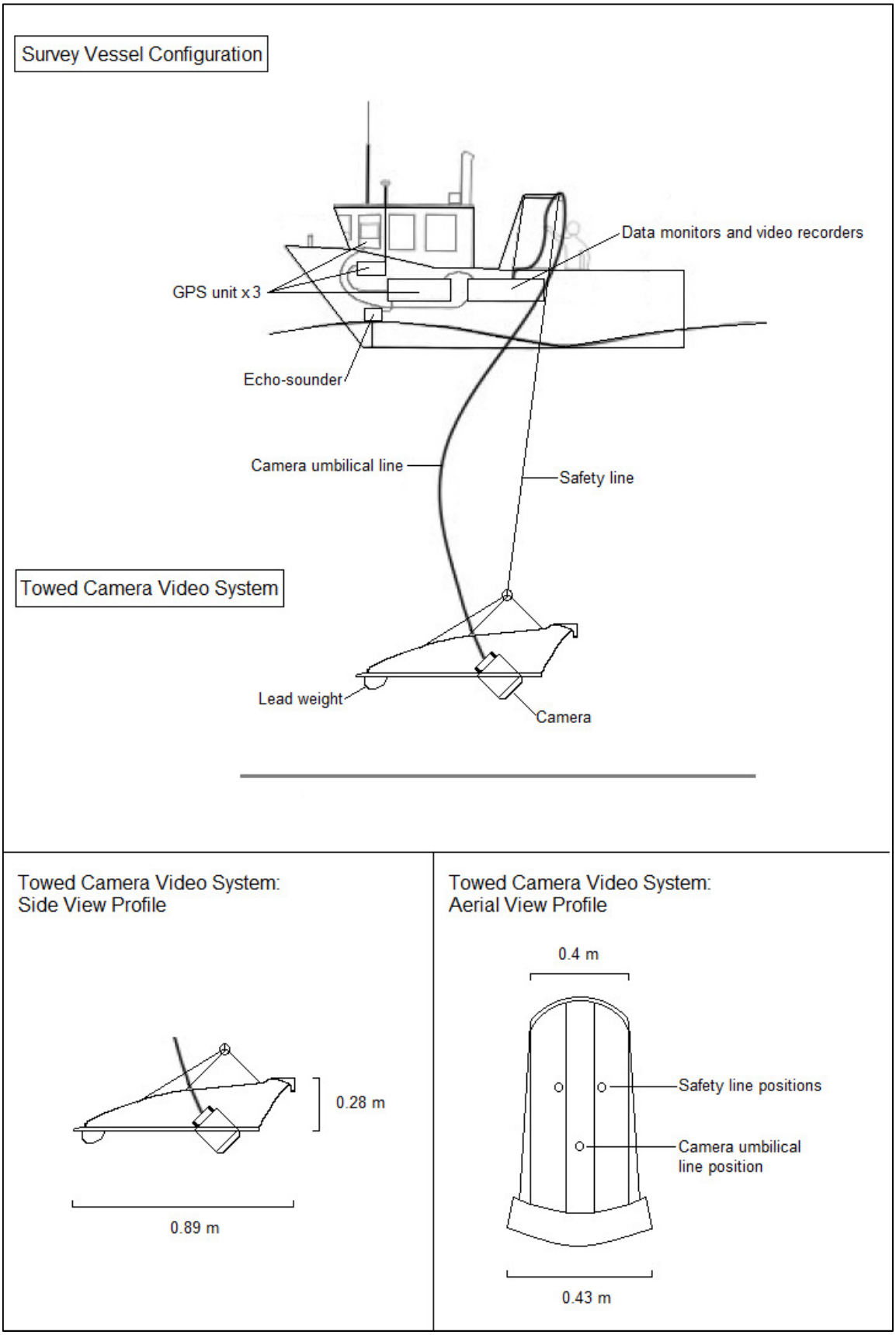


Table A.2.2. Waypoints of the T60 and T61 benthic survey square and sub-survey square transect line positions, Alderney.

Survey square	Transect number	Latitude (Northing)	Longitude (Westing)
T60		49° 42; 500 N	02° 09; 80 W
	1	49° 42; 255 N	02° 09; 79 W
	2	49° 42; 175 N	02° 09; 70 W
	3	49° 42; 125 N	02° 09; 60 W
	4	49° 42; 075 N	02° 09; 50 W
	5	49° 42; 000 N	02° 09; 41 W
T61		49° 42; 500 N	02° 08; 20 W
	1	49° 42; 225 N	02° 08; 79 W
	2	49° 42; 175 N	02° 08; 70 W
	3	49° 42; 125 N	02° 08; 60 W
	4	49° 42; 075 N	02° 08; 50 W
	5	40° 42; 000 N	02° 08; 41 W

Notes: The T60 and T61 survey square positions are based on their general central points, with transect line positions (n = 5) based on their individual starting points for the towed camera video assessment (from dGPS). Transect lines are numbered from their most Northerly position within each square (1 being the most Northerly). All positions are based on WGS84 reference coordinate system.

Table A.2.3. Mean benthic physical and ecological composition identified per temporal sampling period (SP) within the Alderney Race T60 survey square, from 2009 – 2010.

Physical components	SP February 2009		May 2009		August 2009		November 2009		February 2010		May 2010	
	Mean	SD	Mean	SD	Mean	SD	Mean	SD	Mean	SD	Mean	SD
Bare bedrock	5	5.65	15.5	16.7	5.4	8.96	5	8.1	5	5.65	2.5	2
Boulders (grain size 0.25 - 3 m)					3	5.19			3.5	3.53	2	0.7
Cobbles (grain size 64 – 256 mm)			14	10	30.6	11.5	18	13	13	4.24	24	14.1
Pebbles (grain size 4 – 64 mm)	28.5	2.12	4	2								
Gravel (grain size 2 – 4 mm)			3	2.2	7	12.1	24	20	35.2	23.6	43.5	33.2
Sand (grain size < 2 mm)	47	2.82	14.5	12	8	7.54						
Organic material												
Cover species	19.5	6.36	49	47	45.98	27.3	53.5	51	43.3	40.1	28	23
Singleton species	0		1		6		8		4		14	

Table A.2.4. Mean benthic physical and ecological composition identified per temporal sampling period (SP) within the Alderney Race T61 survey square, from 2009 – 2010.

Physical components	SP February 2009		May 2009		August 2009		November 2009		February 2010		May 2010	
	Mean	**SD**	**Mean**	**SD**	**Mean**	**SD**	**Mean**	**SD**	**Mean**	**SD**	**Mean**	**SD**
Bare Bedrock	8	2.82	5	1	19	15.5	1.1	2.1	23.1	23.4	4.6	3.53
Boulder (grain size 0.25 - 3 m)									9.25	8	8.25	1.76
Cobbles (grain size 64 – 256 mm)			7	3.5	5	7.07	1	0.21	12.5	10.2	14	2.82
Pebbles (grain size 4 – 64 mm)	7	4.24	7.5	4								
Gravel (grain size 2 – 4 mm)			2	0.1	3.5	4.94	29.5	17.8	35.2	30	34.2	6.01
Sand (grain size < 2 mm)	35	7.07	11.5	7	29.5	21.9			1	0.2		
Organic material									0.25	0.1		
Cover species	50	14.4	67	58	43	25.4	68.4	40	18.7	6	39	8.48
Singleton species	0		2		4		11		3		12	

Notes: Physical and ecological proportions are given as mean percentage cover (%), with standard deviations (SD) and total abundance counts (singleton species), across all sampling intervals for both survey squares, derived from the towed video camera photographic stills.

Appendix 3. Supporting information for chapter 3.

Table A.3.1. SIMPER (%) results for macroalgae species composition in the upper shoreline height: a) differences between Longis Bay (LB) and Control Bay (CB), b) differences within bays across years and c) differences within bays across seasons.

a) Bay	**LB**	**CB**						
Chlorophyta								
Ulva intestinalis	15.40	10.81						
Phaeophyta								
Ascophyllum nodosum	**31.35**							
Fucus serratus								
Fucus spiralis	12.11	**45.99**						
Fucus vesiculosus	**31.63**	**24.39**						
Pelvetia canaliculata		13.47						
Average similarity	27.79	51.88						
b) Year	**2009**		**2010**		**2011**			
	LB	**CB**	**LB**	**CB**	**LB**	**CB**		
Chlorophyta								
Cladophora rupestris								
Ulva intestinalis	14.61	17.39	14.21	5.59	**28.90**	15.10		
Ulva lactuca		2.58						
Ulva linza								
Phaeophyta								
Ascophyllum nodosum	**33.10**		**33.38**		19.60			
Fucus serratus			14.05					
Fucus spiralis	8.40	**46.29**		**46.86**	10.33	**37.45**		
Fucus vesiculosus	**32.24**	12.41	**28.84**	**25.78**	**39.66**	**39.62**		
Pelvetia canniculata		13.38		14.27				
Rhodophyta								
Osmundea pinnatifida	4.44							
Average similarity	21.87	51.09	23.06	48.67	21.69	58.13		
c) Season	**Feb**		**May**		**August**		**Nov**	
	LB	**CB**	**LB**	**CB**	**LB**	**CB**	**LB**	**CB**
Chlorophyta								
Cladophora rupestris								4.23
Ulva intestinalis	14.55	13.42	**30.42**	**23.76**	12.02		19.89	
Phaeophyta								
Ascophyllum nodosum	**35.12**		**29.66**		**28.43**		**20.07**	6.95
Fucus serratus	7.81						14.38	
Fucus spiralis	6.77	**46.84**	16.11	**44.88**	8.43	**45.17**		**34.08**
Fucus vesiculosus	**25.64**	**26.66**	**22.21**	16.09	**42.59**	**31.16**	**43.70**	**25.36**
Pelvetia canaliculata		6.01		7.48		15.49		**22.56**
Rhodophyta								
Osmundea pinnatifida	5.80							
Average similarity	22.33	52.79	20.61	47.81	29.36	60.10	13.94	45.64

Table A.3.2. SIMPER (%) results for macroalgae species composition in the mid shoreline height: a) differences between Longis Bay (LB) and Control Bay (CB), b) differences within bays across years and c) differences within bays across seasons.

a) Bay	**LB**	**CB**						
Chlorophyta								
Ulva intestinalis	6.80	4.77						
Ulva lactuca	4.53	6.40						
Phaeophyta								
Ascophyllum nodosum	**21.59**							
Fucus serratus	**22.98**	7.88						
Fucus vesiculosus	**35.34**	**62.20**						
Rhodophyta								
Lithophyllum spp		5.60						
Osmundea pinnatifida		6.03						
Average similarity	26.35	55.08						
b) Year	**2009**		**2010**		**2011**			
	LB	**CB**	**LB**	**CB**	**LB**	**CB**		
Chlorophyta								
Cladophora rupestris	6.08		2.58					
Ulva intestinalis		3.64	10.58	7.58	12.78	6.48		
Ulva lactuca	6.27	8.24	7.95	6.57		4.78		
Ulva linza	3.20							
Phaeophyta								
Ascophyllum nodosum	18.26		11.79		**23.06**			
Fucus serratus	**28.22**	6.70	**21.97**	6.41	17.57	11.90		
Fucus vesiculosus	**25.46**	**60.50**	**32.48**	**58.85**	**39.72**	**54**		
Rhodophyta								
Cantenella caespitosa								
Corallina officinalis		4.75						
Lithophyllum spp		5.15	2.82	4.20		7.86		
Osmundea pinnatifida		4.55		9.71		6.85		
Vertebrata lanosa	3.22							
Average similarity	16.60	67.83	23.75	55.18	22.51	61.10		
c) Season	**Feb**		**May**		**August**		**Nov**	
	LB	**CB**	**LB**	**CB**	**LB**	**CB**	**LB**	**CB**
Chlorophyta								
Cladophora rupestris	5.47						4.98	
Ulva intestinalis			**23.46**	4.87	4.10	9.70		7.26
Ulva lactuca	2.57	4.34	9.79	13.49	6.40			7.89
Phaeophyta								
Ascophyllum nodosum	**34.26**		4.50		10.68		**20.83**	
Fucus serratus	16.92	12.28	**22.22**	5.75	18.62	6.89	**33.81**	7.19
Fucus vesiculosus	**23.84**	**44.90**	**33.38**	**55.79**	**43**	**73.93**	**32.35**	**55.55**
Rhodophyta								
Corallina officinalis	2.75	5.39						
Lithophyllum spp	2.56	12.33		5.05				3.95
Osmundea pinnatifida	3.42	12.45		5.63				10.81
Vertebrata lanosa					8.84			
Average similarity	20.19	56.91	25	58.31	15.35	64.61	23.64	67.93

Table A.3.3. SIMPER (%) results for macroalgae species composition in the lower shoreline height: a) differences between Longis Bay (LB) and Control Bay (CB), b) differences within bays across years and c) differences within bays across seasons.

a) Bay	**LB**	**CB**
Chlorophyta		
Ulva intestinalis	11.99	4.34
Ulva lactuca	6.85	4.80
Phaeophyta		
Ascophyllum nodosum	1.46	
Fucus serratus	**38.74**	7.39
Fucus vesiculosus	17.27	**48.66**
Rhodophyta		
Corallina officinalis		4.69
Lithophyllum spp	4.05	12.83
Osmundea pinnatifida	8.09	10.76
Angiosperm		
Zostera marina	2.80	
Average similarity	34.49	54.14

b) Year	**2009**		**2010**		**2011**	
	LB	**CB**	**LB**	**CB**	**LB**	**CB**
Chlorophyta						
Cladophora rupestris	1.69					
Codium fragile subsp *fragile*			2.52			
Ulva intestinalis	11.68	4.42	6.90	3.91	**27.01**	9.96
Ulva lactuca	12.27	8.18	6.62	2.97		2.45
Phaeophyta						
Ascophyllum nodosum	1.91					
Bifucaria bifucata			2.97			
Fucus serratus	**39.77**	6.53	**39.85**	4.90	**25.02**	4.69
Fucus vesiculosus	16.13	**49.32**	8.44	**44.17**	**21.44**	**46.15**
Himanthalia elongata			5.84	2.77		
Rhodophyta						
Corallina officinalis				5.81		4.67
Lithophyllum spp		11.98	9.47	14.53	6.67	11.94
Osmundea pinnatifida	7.36	10.09	8.36	13.60	10.59	11.44
Average similarity	39.34	56.06	35.76	55.83	35.06	62.03

c) Season	**Feb**		**May**		**August**		**Nov**	
	LB	**CB**	**LB**	**CB**	**LB**	**CB**	**LB**	**CB**
Chlorophyta								
Codium fragile subsp *fragile*	2.87				2.07			
Ulva intestinalis	10.81		**22.98**	9.60	12.31	10.57	8.54	
Ulva lactuca		5.62	10.34	4.20	8.78	5.56	8.54	
Phaeophyta								
Ascophyllum nodosum					1.61			
Bifucaria bifucata	2.25							3.04
Fucus serratus	**26.45**		**37.44**	4.01	**38.23**	10.90	**41.12**	

c) Season	**Feb**		**May**		**August**		**Nov**	
	LB	**CB**	**LB**	**CB**	**LB**	**CB**	**LB**	**CB**
Fucus vesiculosus	16.13	**40.90**	13.24	**54.08**	14.89	**42.55**	15.15	**49.62**
Himanthalia elongata					5.94	2.73		
Rhodophyta								
Ahnfeltia plicata	3.19							
Chondrus crispus						4.24		
Corallina officinalis	3.17	8.34		4.40				3.94
Lithophyllum spp	12.66	19.19		8.36	3.68	7.08	6.74	17.49
Osmundea pinnatifida	13.68	17.36	9.81	7.19		6.65	12.20	16.40
Rhodothamniella floridula					3.20			
Average similarity	30.25	57.72	34.88	55.30	42.91	54.70	40.73	65.24

Table A.3.4. SIMPER (%) results for faunal species composition in the upper shoreline height: a) differences between Longis Bay (LB) and Control Bay (CB), b) differences within bays across years and c) differences within bays across seasons.

a) Bay	**LB**	**CB**						
Mollusca								
Gibbula cineraria	6.04	4.52						
Gibbula umbilicalis		4.35						
Littorina obtusata	19.78							
Osilinus lineatus	**28.64**	**47.83**						
Patella vulgata	**41.23**	**35.07**						
Average similarity	15.67	48.14						
b) Year	**2009**		**2010**		**2011**			
	LB	**CB**	**LB**	**CB**	**LB**	**CB**		
Mollusca								
Gibbula cineraria		8.62	9.78	8.38				
Gibbula umbilicalis				16.56	12.15			
Littorina littorea					21.14	13.20		
Littorina obtusata	**34.27**		**30.90**					
Littorina saxatilis	12.90	17.52						
Osilinus lineatus	**27.29**	**45.32**		**32.21**	**33.81**	**55.38**		
Patella vulgata	17.40	**21.77**	**53.99**	**40.82**	**32.90**	**25.55**		
Average similarity	17.27	57.93	20.43	58.33	11.70	58.86		
c) Season	**Feb**		**May**		**August**		**Nov**	
	LB	**CB**	**LB**	**CB**	**LB**	**CB**	**LB**	**CB**
Mollusca								
Gibbula cineraria	5.37		17.03	14.07		6.52		
Gibbula umbilicalis	5.65	19.63						
Littorina littorea	9.84	9.35						
Littorina obtusata	**28.68**						**78.67**	
Littorina saxatilis			**23.58**	7.25		11.63		
Osilinus lineatus	5.49	**22.42**	**29.90**	**49.49**	**35.85**	**52.92**	11.87	**53.07**
Patella vulgata	**37.98**	**40.48**	**29.49**	**19.54**	**56.87**	**24.02**		**38.89**
Average similarity	22.35	61.05	9.54	57.23	17.53	63.69	17.57	47.85

Table A.3.5. SIMPER (%) results for faunal species composition in the mid shoreline height: a) differences between Longis Bay (LB) and Control Bay (CB), b) differences within bays across years and c) differences within bays across seasons.

a) Bay	**LB**	**CB**						
Mollusca								
Gibbula cineraria	14.46	8.51						
Gibbula umbilicalis	3.87	8.36						
Osilinus lineatus	15.68	**29.79**						
Patella vulgata	**57.61**	**47.59**						
Average similarity	23.55	63.05						
b) Year	**2009**		**2010**		**2011**			
	LB	**CB**	**LB**	**CB**	**LB**	**CB**		
Mollusca								
Gibbula cineraria	17.53	**20.92**	**24.35**	8.69				
Gibbula pennanti	14.49							
Gibbula umbilicalis		3.47	4.58	12.45		15.01		
Littorina saxatilis	6.96							
Osilinus lineatus	10.48	29.93	5.98	**22.80**	**35.81**	**33.20**		
Patella depressa		3.82						
Patella vulgata	**41.88**	**33.81**	**59.45**	**49.52**	**57.78**	**46.42**		
Average similarity	20.53	69.54	44.04	69.11	13.40	71.28		
c) Season	**Feb**		**May**		**August**		**Nov**	
	LB	**CB**	**LB**	**CB**	**LB**	**CB**	**LB**	**CB**
Cnidaria								
Actinia equina	3.61						12.93	
Mollusca								
Gibbula cineraria	17.15	14.57	**22.36**	15.43	**23.11**	5.16	9.81	7.53
Gibbula pennanti			12.30					
Gibbula umbilicalis		6.48		5.70	3.14	10.14	13.72	19.93
Osilinus lineatus		**28.13**	10.39	**23.53**	17.54	**35.49**	16.02	**24.94**
Patella depressa		3.53						
Patella vulgata	**70.61**	**38.94**	**47.43**	**47.29**	**48.24**	**43.08**	**44.35**	**42.58**
Average similarity	30.25	69.29	27.94	67.33	29.18	69.05	19.40	75.71

Table A.3.6. SIMPER (%) results for faunal species composition in the lower shoreline height: a) differences between Longis Bay (LB) and Control Bay (CB), b) differences within bays across years and c) differences within bays across seasons.

a) Bay	**LB**	**CB**						
Cnidaria								
Actinia equina		6.34						
Mollusca								
Gibbula cineraria	7.88	6.47						
Gibbula umbilicalis	6.04	9.01						
Osilinus lineatus	11.09	17.19						
Patella vulgata	**68.08**	**56.41**						
Average similarity	44.83	61.12						
b) Year	**2009**		**2010**		**2011**			
	LB	**CB**	**LB**	**CB**	**LB**	**CB**		
Cnidaria								
Actinia equina	3.96			6.12				
Mollusca								
Gibbula cineraria	13.66	10.93	16.25	6.58				
Gibbula umbilicalis			8.99	16.90	13.48	19.56		
Osilinus lineatus	11.90	17.49		11.07	**22.27**	19.12		
Patella depressa	12.60	**23.46**						
Patella vulgata	**51.31**	**40.97**	**68.33**	**55.19**	**61.13**	**55.19**		
Average similarity	40.11	67.14	59.68	71.76	53.98	80.59		
c) Season	**Feb**		**May**		**August**		**Nov**	
	LB	**CB**	**LB**	**CB**	**LB**	**CB**	**LB**	**CB**
Cnidaria								
Actinia equina	4.25							6.52
Mollusca								
Gibbula cineraria	3.59	7.79	**23.68**	6.20		6.50	17.65	
Gibbula umbilicalis	7.80	15.47		6.61	7.83	11.26	11.92	15.58
Osilinus lineatus		15.45	7.25	10.64	**24.36**	19.34	6.83	17.29
Patella depressa	3.27	9.90		10.20	8.66	9.10		
Patella vulgata	**73.86**	**43.21**	**61.16**	**57.56**	**51.12**	**48.64**	**57.57**	**53.71**
Average similarity	55	72.01	44.87	68.15	53.30	76.02	50.79	74.42

Notes: Results are based on separate one-way SIMPER species percentage contribution routines generated from macroalgae percentage cover and faunal abundance count transect datasets. This includes separate analyses of the faunal count datasets; between bays (SIMPER factor: bay) and within each bay (SIMPER factor: year and season). Av Sim represents the similarity of the samples within each factor level. All routines are run in Primer v6, with data square root transformed. Routines are based on the Bray Curtis similarity resemblance measure, with a 90% cut-off selected in all routines. Bold indicating proportions > 20 % SIMPER species contributions.

Appendix 4. Supporting information for chapter 5.

Table A.4.1. Waypoints of the device and control coastline string transect line positions, Isle of Eday.

Survey site	Sting transect line number	Latitude (Northing)	Longitude (Westing)
Device		59° 09; 448 N	02° 49; 561 W
	1	59° 09; 535 N	02° 49; 200 W
	2	59° 19; 550 N	02° 49; 050 W
	3	59° 09; 575 N	02° 48; 850 W
Control			
	1	59° 07; 450 N	02° 46; 700 W
	2	49° 07; 845 N	02° 46; 700 W
	3	49° 07; 945 N	02° 46; 700 W

Notes: String transect line survey site locations are based on their general starting survey points, with all positions are based on WGS84 reference coordinate system, recorded by dGPS systems. String transect lines within the device survey site are positioned at a 171° angle, ran in a South direction. String transect lines within the control survey site are positioned at a 258° angle, ran in an East direction.

Table A.4.2. SIMPER percentage contribution (%) results for benthic species composition a), taxonomic composition b), and feeding regime composition c), across string transect lines within the device and control survey sites, from 2009 – 2010, Isle of Eday.

	Location: Device			**Control**		
	String number			**String number**		
	1	**2**	**3**	**1**	**2**	**3**
a) Species						
Crustacea						
Cancer pagurus	**30.73**	**32.52**	**32.50**	**44.57**	**46.72**	**47.55**
Homarus gammarus	12.65	9.94	11.01			
Necora puber	**34.27**	**37.62**	**39.87**	**51.22**	**53.28**	**52.45**
Echinodermata						
Asterias rubens	10.35	10.53	11.01			
Mollusca						
Buccinum undatum	4.59					
Average similarity	65.97	70.51	67.07	79.89	84.97	78.85
b) Taxon						
Chordata	14.24	5.56				
Crustacea	**58.90**	**76.22**	**80.01**	**88.41**	**100**	**100**
Echinodermata	**20.34**	18.22	19.99	5.29		
Mollusca	6.52			6.30		
Average similarity	86.04	77.15	78.81	81.27	83.39	79.45
c) Feeding regime						
Omnivore	**73.01**	**75.91**	**79.28**	**92.21**	**100**	**93.77**
Predator	**26.99**	**24.09**	**20.72**	7.79		6.23
Average similarity	85.53	90.77	84.93	88.67	87.77	84.72

Notes: Results are based on separate one-way SIMPER percentage contribution routines generated from the benthic assemblages across all string transect line datasets. This includes separate analyses of the survey site datasets for species, taxonomic group and feeding regime type; between the string transect lines (SIMPER factor: string line). Average similarity represents the similarity of the samples within each factor level. All routines are run in Primer v6, with data fourth root transformed. Routines are based on the Bray Curtis similarity resemblance measure, with a 90% cut-off selected for the species datasets and 100% for the group datasets for both survey sites in all routines. Bold indicating proportions ≥ 20 % SIMPER species contributions.

Appendix 5. Supporting information for chapter 6.

Table A.5.1. Locations of the device and control coastline bay sites, Isle of Eday.

Coastline	Bay number	Latitude (Northing)	Longitude (Westing)
Device		59° 09; 950 N	02° 48; 400 W
	1	59° 09; 950 N	02° 48; 300 W
	2	59° 10; 050 N	02° 48; 700 W
	3	59° 09; 700 N	02° 48; 900 W
Control		59° 11; 550 N	02° 45; 900 W
	1	59° 11; 500 N	02° 45; 750 W
	2	49° 11; 200 N	02° 45; 600 W
	3	49° 11; 650 N	02° 45; 500 W

Notes: Coastline and bay site locations are based on their general central points recorded by dGPS, with all positions are based on WGS84 reference coordinate system.

Table A.5.2. Intertidal upper shoreline macroalgae species composition a), taxonomic composition b), and morphological form composition c), SIMPER percentage contributions (%) across the device and control (cont) coastlines for each temporal seasonal sampling period from 2009 – 2010, Isle of Eday.

	Season							
	Oct 2009		**April 2010**		**July 2010**		**Oct 2010**	
	Device	**Cont**	**Device**	**Cont**	**Device**	**Cont**	**Device**	**Cont**
a) Species								
Chlorophyta								
Blindinga minima				4.20				
Cladophora rupestris	8.40		8.08	2.68	8.93	1.76	9.78	2.58
Codium fragile subsp *fragile*	2.68				2.52		7.50	
Ulva intestinalis	8.71	5.55	16.57	10.71	11.10	15.88	8.15	13.38
Ulva lactuca				6.80			1.58	
Phaeophyta								
Ascophyllum nodosum	**18.57**	**28.19**	18.90	**24.81**	19.11	**25.25**	16.47	17.28
Fucus serratus	8.72		3.75		6.29		4.46	4.28
Fucus spiralis	12.40	**26.60**	7.33	**30.49**	**24.12**	**29.95**	14.60	**24.78**
Fucus vesiculosus	**20.67**	19.49	**20.78**	17.09	**23.71**	19.40	16.17	**20.97**
Pelvetia canaliculata	4.66	7.43	11.63			3.61	4.24	3.36
Rhodophyta								
Lithophyllum spp	6.80		7.65				5.74	2.96
Osmundea pinnatifida			1.67				1.87	
Porphyra umbilicalis	1.84						2.07	
Vertebrata lanosa	6.55	12.74	3.65	3.24	4.22	4.16	7.37	10.41
Average similarity	69.55	71.49	72.44	75.09	69.58	72.54	74.76	76.76
b) Taxon								
Chlorophyta	**20.69**	9.19	**30.43**	**25.15**	**24.86**	**29.51**	**26.06**	**24.52**
Phaeophyta	**57.06**	**68.75**	**48.02**	**68.82**	**66.94**	**63**	**51.60**	**55.48**
Rhodophyta	**22.25**	22.06	**21.55**	6.03	8.20	7.49	**22.33**	**20**
Average similarity	87.91	84.42	90.16	83.77	84.50	85.29	95.55	90.51
c) Morphological form								
Filamentous	19.02	9.07	**29.78**	**21.89**	**24.90**	**29.60**	**22.29**	**23.71**
Filamentous (2)	11.88	**22.11**	6.86	5.41	7.89	6.80	13.89	17.85
Foliose single	4.35			11.31			4.19	
Corticated macrophyte			2.83				4.43	
Leathery macrophyte	**52.50**	**68.82**	**47.28**	**61.39**	**67.22**	**63.60**	**44.38**	**53.46**
Articulated calcareous								
Crustose	12.26		13.25				10.82	4.98
Average similarity	75	82.09	75.55	83.64	77.82	80.63	82.52	85.52

Table A.5.3. Intertidal lower shoreline macroalgae species composition a), taxonomic composition b), and morphological form composition c), SIMPER percentage contributions (%) across the device and control (cont) coastlines for each temporal seasonal sampling period from 2009 – 2010, Isle of Eday.

	Season							
	Oct 2009		**April 2010**		**July 2010**		**Oct 2010**	
	Device	**Cont**	**Device**	**Cont**	**Device**	**Cont**	**Device**	**Cont**
a) Species								
Chlorophyta								
Cladophora rupestris	12.23	5.53	5.24	4.31	8.93	2.11	9.13	1.83
Codium fragile subsp *fragile*			4.52		2.47			
Ulva intestinalis	2.93	9.15	4.52	6.68	7.75	3.16	7.34	7.64
Ulva lactuca	1.57		5.76	2.31	1.95	9.38	2.45	
Phaeophyta								
Ascophyllum nodosum	9.36	**21.36**	1.80	16.61	5.97	**22.73**		19.88
Fucus serratus	**25.48**	**23.79**	16.57	16.11	**21.16**	**22.82**	**25.09**	**25.28**
Fucus spiralis	2.90							
Fucus vesiculosus	**23.58**	16.36	12.86	14.48	18.63	19.64	16.61	17.17
Himanthalia elongata			9.87	3.95	3.09		3.54	3.96
Laminaria digitata			11.92	12.25	17.41		9.18	3.08
Pelvetia canaliculata								
Rhodophyta								
Chondrus crispus	3.08	3.03	2	2.06	2.13			
Corallina officinalis			1.78				1.86	
Lithophyllum spp	7.20	8.14	8.44	8.51	1.57	7.07	12.35	3.82
Osmundea pinnatifida	2.84	2.14	10.59	4.84	2.35	3.38	12.45	9.65
Vertebrata lanosa	8.82	10.51	4.12	7.88	6.59	9.72		7.68
Average similarity	61.08	82.07	82.53	80.71	69.19	71.14	67.22	74.84
b) Taxon								
Chlorophyta	**21.64**	18.44	19.76	19.64	**23.66**	19.05	18.86	16.45
Phaeophyta	**54.21**	**57.48**	**51.53**	**55.31**	**59.11**	**59.85**	**51.70**	**58.33**
Rhodophyta	**24.15**	**24.09**	**28.70**	**25.05**	17.23	**21.10**	**29.44**	**25.22**
Average similarity	89.85	89.21	96.54	95.45	89.52	89.96	92.24	90.45
c) Morphological form								
Filamentous	17.70	15.52	12.72	13.14	**21.46**		14.17	**15.84**
Filamentous (2)	13.59	17.21	6.87	14.50	12.65	14.81		14.32
Foliose single	2.03		9.63	4.06	3.49	14.28	10.01	
Corticated macrophyte	12.07	5.40	17.69	12.17	4.78	5.60	17.01	6.71
Leathery macrophyte	**44.74**	**48.54**	**36.01**	**40.47**	**54.81**	**49.44**	**39.32**	**56.01**
Articulated calcareous			2.97				2.62	
Crustose	9.89	13.34	14.11	15.66	2.81	10.79	16.86	7.12
Average similarity	80.58	81.82	91.39	86.34	78.16	80.53	84.60	77.15

Table A.5.4. Intertidal upper shoreline faunal species composition a), taxonomic composition b), and feeding regime composition c), SIMPER percentage contributions (%) across the device and control (cont) coastlines for each temporal seasonal sampling period from 2009 – 2010, Isle of Eday.

	Season							
	Oct 2009		**April 2010**		**July 2010**		**Oct 2010**	
	Device	**Cont**	**Device**	**Cont**	**Device**	**Cont**	**Device**	**Cont**
a) Species								
Cnidaria								
Actinia equina			**30.58**					
Mollusca								
Gibbula cineraria								
Littorina littorea				**23.61**	**40.94**	**100**		
Littorina obtusata			6.81	**51.72**			**29.19**	**50.29**
Littorina saxatilis		**78.88**					6.88	
Nucella lapillus	**38.13**				8			
Patella depressa							8.18	
Patella vulgata	**61.87**	**21.12**	**62.60**	**24.67**	**51.05**		**55.74**	**49.71**
Average similarity	46.17	57.71	62.08	50.02	71.97	36.98	64.85	58.46
b) Taxon								
Cnidaria			**32.39**					
Mollusca	**100**	**100**	**67.61**	**100**	**100**	**100**	**100**	**100**
Average similarity	66.67	82.04	92.62	89.48	92.21	84.71	84.32	80.52
c) Feeding regime								
Epifaunal herbivore	**64.07**	**100**	**66.28**	**100**	**87.50**	**100**	**100**	**100**
Predator	**35.93**		**33.72**		**12.50**			
Average similarity	78.82	67.28	92.63	89.48	80.15	70.50	84.32	77.77

Table A.5.5. Intertidal lower shoreline faunal species composition a), taxonomic composition b), and functional form composition c), SIMPER percentage contributions (%) across the device and control (cont) coastlines for each temporal seasonal sampling period from 2009 – 2010, Isle of Eday.

	Season							
	Oct 2009		**April 2010**		**July 2010**		**Oct 2010**	
	Device	**Cont**	**Device**	**Cont**	**Device**	**Cont**	**Device**	**Cont**
a) Species								
Cnidaria								
Actinia equina	**26.21**	7.18	17.01	16.39	**21.86**	**32.63**	19.38	**22.68**
Mollusca								
Gibbula cineraria			4.15					
Littorina littorea					13.22	9.20	12.55	
Littorina obtusata				16.39				
Littorina saxatilis		**25.58**					5.66	
Nucella lapillus	16.80	**27.86**	18.14	8.22	**25.90**	10.65	**22.34**	9.99
Patella depressa	**26.39**		**20.53**	**27.48**		8.96		**29.53**
Patella vulgata	**30.60**	**39.38**	**40.17**	**31.52**	**39.01**	**38.57**	**40.07**	**37.81**
Average similarity	73.89	67.76	77.48	70.56	78.81	51.83	73.13	62.51
b) Taxon								
Cnidaria	**38.97**	**11.52**	**27.62**	**30.30**	**33.17**	**41.09**	**31.50**	**34.42**
Mollusca	**61.03**	**88.48**	**72.08**	**69.70**	**66.83**	**58.91**	**68.50**	**65.58**
Average similarity	89.87	78.37	96.53	85.89	90.30	75.47	92.38	76.85
c) Feeding regime								
Epifaunal herbivore	**56.60**	**64.38**	**67.14**	**63.01**	**59.46**	**57.08**	**61.91**	**62.33**
Predator	**43.40**	**35.62**	**32.86**	**36.99**	**40.54**	**42.92**	**38.09**	**37.67**
Average similarity	88.95	87.95	96.65	84.41	90.34	74.51	92.34	73.83

Notes: All intertidal macroalgae and faunal results are based on separate one-way SIMPER analyses. Bold indicates ≥ 20 %, with average within-group similarities shown. All analyses are based on fourth-root transformations.