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NOVEL METHODS OF BENTHIC HABITAT ASSESSMENT IN DESIGNATED WATERBODIES AROUND IRELAND

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A thesis submitted to the department of Zoology, School of Natural Sciences, National University of Ireland, Galway, in fulfilment of the degree of Doctor of Philosophy, January 2017.

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Declaration

I certify that this thesis which I now submit for examination for the award of PhD, is entirely my own work and has not been taken from the work of others save and to the extent that such work has been cited and acknowledged within the text of my work.

This thesis was prepared according to the regulations for postgraduate study by research of the National University of Ireland, Galway and has not been submitted in whole or in part for an award in any other Institute or University.

The work reported on in this thesis conforms to the principles and requirements of the University's guidelines for research.

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Abstract

The European Union has implemented some of the most progressive conservation Directives in the world. These Directives are implemented in different ways, have different reporting cycles and use unique terminologies. The objectives of all the Directives require that the ecosystem based approach be applied when assessing the ‘status’ of the benthic environment. Benthic ecologists are faced with particular challenges when addressing the requirements of these Directives.

Some anthropogenic activities in designated waterbodies can be difficult to monitor due to the variable nature of the environment in which they occur. Some of the newest forms of anthropogenic activities in the coastal environment may have significant impacts on local hydrographic regimes. This will require the relationship between the benthos and modified hydrodynamic conditions to be assessed. Large scale predictive models will be required to form baseline datasets for benthic communities within Ireland’s Exclusive Economic Zone (EEZ). Large scale mapping initiatives such as these are financially costly and it is important for benthic ecologists to understand the effectiveness of the available tools and resources prior to carrying out such studies.

This thesis presents some new methods of benthic habitat assessment which can help to circumvent some of the key issues facing benthic ecologists working in response to EU Directives. We assess the effects of intertidal bivalve trestle cultivation on important bird feeding grounds in Natura 2000 sites. We present the Ecological Quality Ratio (EQR) Infaunal Quality Index (IQI) as an effective tool for assessing the risk posed to the conservation status of these sites.

The potential effects of tidal energy extraction on epibenthic reef communities in a Natura 2000 site are assessed. Case studies are presented that assess the spatial and temporal effects of tidal turbine installation and operation. The relationship between epibenthic communities and a range of current velocities required to produce rated tidal energy are assessed. The turbulent wake of a tidal energy turbine was simulated and the relationship between epibenthic community structure and the simulated modified wake is assessed. We present the High Energy Hard Substrate (HEHS) index, a new EQR that was developed for use in animal-dominated, stable reefs, as an effective monitoring tool within tidal energy extraction sites.

The effectiveness of hydrodynamic modelling and acoustically classified sediment data, as predictors of benthic biotopes was assessed using Galway Bay as a case study. This approach has potential application in mapping Irelands EEZ for the purposes of the Marine Strategy Framework Directive (MSFD).

1. Introduction

The European Union has implemented some of the most progressive legislation on ecosystem-based approaches to marine conservation in the world (Borja et al., 2010b). Marine conservation of European seas is currently addressed by four Directives; the Habitats Directive (HD: Council Directive 92/43/EEC); the Birds Directive (BD; Council Directive 2009/147/EC); the Water Framework Directive (WFD: Council Directive, 2000/60/EC); the Marine Strategy Framework Directive (MSFD; Council Directive 2008/56/EC).

Each Directive pursues its conservation objectives through unique implementation strategies and reporting structures, and in doing so use different terminologies. Their conservation objectives range from the removal of pollution from the marine environment (WFD and MSFD), to the protection of single species or large scale habitats (HD, BD, MSFD). The ultimate goal of the Directives is the conservation, and when necessary, restoration of environmental integrity and biodiversity. This shared objective is enough to make integration and mutual enlightenment between them an obtainable goal. For benthic habitat assessments, integration of the Directives would require the establishment of best practices and standardised outputs, so that conservation objectives and monitoring outputs are comparable across the Directives.

New methods of benthic habitat assessment that are based on pre-existing resources and readily available resources are proposed through six case studies. It is the aim of this thesis to introduce solutions to some of the key issues faced by benthic habitat mapping and applied impact studies carried out in response to current EU Directives.

1.1 The Directives

The HD established a framework for the protection and improvement of marine, freshwater and terrestrial environments, through the conservation of natural habitats and their resident wild flora and fauna. The directive is aimed at promoting biodiversity while still taking into account economic, social, cultural and regional requirements of the European Community. Habitats and their resident flora and fauna are part of the natural heritage of the Community, and are frequently under threat from human activities. Habitats and species that are deemed to be of high conservation value are to be reported as being of priority, which allows conservation measures to be implemented, principally, through the designation of the Sites as Special Areas of Conservation (SACs) (Council Directive 92/43/EEC).

The HD is pursuant to the BD which established a similar Europe wide policy for the protection of bird species and their habitats. Many European bird species are recognised as being threatened and in rapid decline in some cases (Council Directive 79 / 409 / EEC). Article 3 of the HD requires that a 'coherent European ecological network of special areas of conservation is established'. Specially Protected Areas (SPAs) are designated for the protection of birds, as well as all SACs, and are incorporated into the European ecological network, Natura 2000 (http://ec.europa.eu/environment/nature/natura2000).

Natura 2000 sites are designated not for the total exclusion of human activities, but for the sustainable management of local economies at no, or at least minimal, ecological expense. These sites are widespread throughout Europe, covering 18% of terrestrial, and 6% of marine territories (http://ec.europa.eu/environment/nature/natura2000/index_en.htm).
Frequently, human activities come in direct contact with Natura 2000 sites by either occurring within or immediately adjacent to them. Under article 6(3) of the HD directive, any human activity not directly connected to the conservation of the species or habitat present must undergo appropriate assessment. When a plan or project has gained state and public approval, appropriate assessments helps to maintain the 'Favourable Conservation Status' (FCS: See Fig. 1) of a designated habitat by monitoring the potential impacts associated with the project. Only in exceptional cases do proposed projects gain permission to continue development despite significantly affecting a site's conservation status. These cases arise when the project is deemed to be of 'overriding public interest'.

The WFD provides a framework for the improvement and protection of inland ground and surface waters as well as transitional and coastal waters within all European Union (EU) member states. The WFD was established to protect Europe's aquatic ecosystems from pollution and requires all EU member states to maintain, and when necessary improve, the quality of all inland ground and surface waters as well as transitional and coastal waters. The protection of inland water catchments, ground and surface waters, has subsequent positive benefits for terrestrial ecosystems drawing on groundwater supplies, aquatic ecosystem quality in rivers, estuaries and the coastal environment (Borja et al., 2003).

The final objective is for all water bodies to achieve at least an Ecological Status (ES: See Table. 1) of ‘Good’ or higher by 2015. To achieve this, each member state is required to assess the ES of its water bodies and where necessary establish management plans to restore or maintain good ES. The ES is assigned using an integrative approach involving the assessment of hydromorphological, physicochemical and biological quality elements, with particular emphasis placed on the biological component in the transitional and coastal environment (Borja and Muxika, 2005).

The MSFD is a framework that requires all member states to take the necessary measures to achieve or maintain Good Environmental Status (GES: See Table. 1) in the marine environment by 2020 (Barry et al., 2013). Under the MSFD, EU Member States are required to determine a set of characteristics for GES on the basis of eleven Quality Descriptors, each addressing a critical component of the ocean ecosystem or a form of pertinent human impact. One of the key components of the ocean ecosystem is seafloor habitat integrity, which must be “at a level that ensures the structure and functions of ecosystems are safeguarded and benthic ecosystems, in particular, are not adversely affected”. Strategies devised by member states in response to the MSFD must apply an ecosystem approach to the management of human activities involving the use of marine goods and services. The establishment of Marine Protected Areas (MPAs) will be integral to halting marine biodiversity loss and in maintaining GES (Barry et al., 2013). An area is said to have a GES if its environment is being used at a sustainable level. An area with GES can be expected to host clean waters that are ecologically diverse, healthy and productive within their intrinsic conditions. The MSFD assessment area covers 490,000 km^2 of Irish waters within the boundaries of Ireland’s Exclusive Economic Zone (EEZ). This incorporates coastal waters, except those designated under the WFD, to abyssal plains at depths up to 5000 m. Transitional waters are not included.

The MSFD is an all-encompassing directive, it has similarities to Natura 2000 in that it emphasises the importance of designating protected areas. It also shares commonalities with the WFD, as it too addresses the need for pollution monitoring in conjunction with the designation of areas for constant monitoring. The MSFD includes a requirement to adopt specific standardised methods to ensure consistency and comparability of its implementation across Europe. This ideology should be adopted when considering benthic habitat assessments carried out in response to all of the Directives (Borja et al., 2010b).

1.2 Standardising benthic monitoring outputs across the Directives

The common goal permeating the directives is the protection and sustainable management of the marine environment using an ecosystem based approach. Each has a different terminology that applies to this shared goal (See Table. 1). In each case, the directives refer to an environmental 'Status', that when acceptable, the habitat in question is healthy, functional and in sustainable equilibrium.

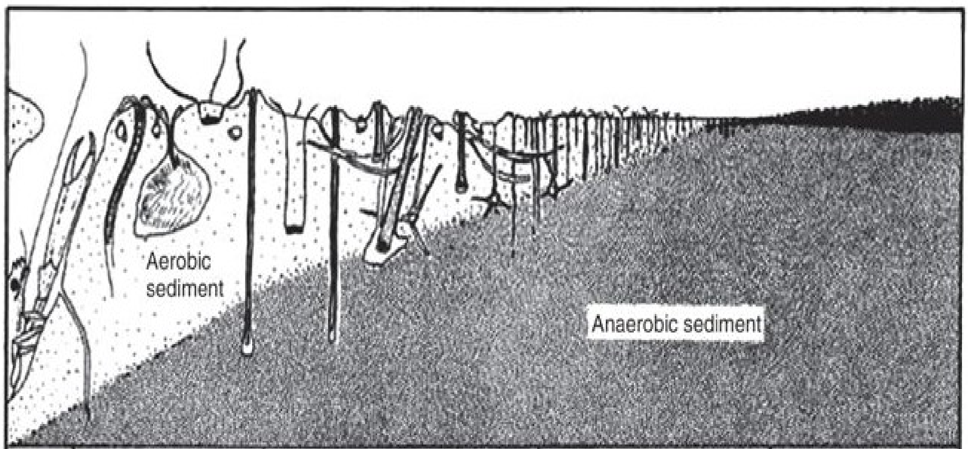
Conservation Goal Definitions
Favourable Conservation Status (HD and BD) — *"its natural range and areas it covers within that range are stable or increasing, and the specific structure and functions which are necessary for its long-term maintenance exist and are likely to continue to exist for the foreseeable future"*
Good Environmental Status (MSFD) — *"the environmental status of marine waters where these provide ecologically diverse and dynamic oceans and seas which are clean, healthy and productive within their intrinsic conditions, and the use of the marine environment is at a level that is sustainable, thus safeguarding the potential for uses and activities by current and future generations"*
Good Ecological Status (WFD) — "*the values of the biological quality elements for the surface water body type show low levels of distortion resulting from human activity, but deviate only slightly from those normally associated with the surface water body type under undisturbed conditions."*

Table 1: List of definition of the conservation objectives for the Directives

1. Introduction

The recognition of this shared goal is reflected in the attempts at integrating the WFD, MSFD, HD and BD by government agencies within member states (Borja et al., 2010, European Economic Interest Group, 2015) Central to the integration of the directives is identifying common indicators of the status of the environment in each case. Conveniently, within the benthic environment, in-particular the sedimentary benthic environment, there is a large body of literature regarding how macrofaunal distributions are indicative of environmental conditions (Wass, 1967, Dauer, 1993, Tapp et al., 1993, Pearson and Rosenberg, 1978, Borja et al., 2000).

Pearson and Rosenberg (1978) developed a successional model that describes how a subtidal sedimentary community changes in time and space relative to organic enrichment. This model is based on facilitation (Connell and Slatyer, 1977) which describes how stress tolerant opportunistic taxa, through their mechanistic ecological roles, create niche space for less stress tolerant taxa.



Zone	Normal	Transitory		Polluted	Grossly polluted
Typical macrofauna dominant	*Nucula* *Amphiura* *Torobelllides* *Rhodine* *Echinocardium* *Nephrops*	*Labidoplax* *Corbula* *Goniada* *Thyasira* *Pholoe*	*Chaetozone* *Anaitides* *Poctinaria* *Myriocheie* *Ophiodromus*	*Capitella* *Scoloiepis*	No macrofauna Surface covered By fiber 'blanket'

Figure 1: Benthic community response to pollution disturbance. Source; Pearson and Rosenberg (1978).

According to the Pearson and Rosenberg (1978) model, with increasing proximity to a disturbed area a reduction in biodiversity, abundance and biomass of macrofauna occurs. Stress sensitive, large-bodied, infaunal deposit feeders are also increasingly replaced by small-bodied opportunistic taxa. This effect on community structure dissipates in a predictable manner with spatial and temporal distance from a disturbance source.

Multimetric indices developed in response to the WFD have employed this paradigm of using the functional roles of animals to make quantitative measurements of pollution on the marine environment. ES is assigned through the assessment of water bodies based on biological, hydromorphological and physicochemical quality elements. An Ecological Quality Ratio (EQR) is derived by comparing putatively disturbed conditions to reference undisturbed conditions (Borja et al., 2000, Borja et al., 2007). EQRs have a decimal value between 0 and 1. Values close to 1 indicate a “High” ES and values close to 0 indicate an ES of “Bad”. There are five ES classes (Bad, Poor, Moderate, Good, High) each represents a range of values that exist between class boundary cut off points along the full EQR value range. For the purposes of the WFD, EQRs must include metrics that address ‘the level of diversity and abundance of invertebrate taxa’ and ‘the proportion of disturbance sensitive taxa’ (Borja et al., 2007). The successional model developed by Pearson and Rosenberg (1978) is central to many benthic EQRs used in soft subtidal sediments (Borja et al., 2000, Prior, 2004, Borja et al., 2007, Muxika et al., 2007, Mackie, 2009, Phillips et al., 2014). The “Moderate”/ “Good” boundary is critical under the WFD. If an area is assigned an ES of Moderate or less, remedial management action is required to improve the ES of the area to ‘Good’ or higher.

Benthic EQRs such as the Infaunal Quality Index (IQI) were initially developed for use in low energy sub-tidal soft sediments. However, they have been shown to be flexible in their application in different habitat types (Van Hoey et al., 2007, Fitch and Crowe, 2010, Kennedy et al., 2011, Forde et al., 2013, Fitch et al., 2014, Forde et al., 2015), across large geographical distances (Forde et al., 2013), amenable to modification (Forde et al., 2013, Forde et al., 2015) and robust to changes in sampling methodologies (Kennedy et al., 2011). Multimetric and biotic indices have been used to monitor a multitude of anthropogenic impacts in the marine environment, these include; sewerage, wastewater and industrial outfalls (Borja et al., 2000, Borja et al., 2003), the disposal of waste products from oil drilling such as ester based drilling muds (Borja et al., 2003, Muxika et al., 2005), sand extraction (Van Dalfsen et al., 2000), direct dredging impacts and the subsequent disposal of dredge waste (Rhoads et al., 1978), the deposition of mining debris, the construction of marinas (Muxika et al. 2005) and various modes of aquaculture (Bouchet and Sauriau, 2008, Borja et al., 2009a, Forde et al., 2015). The same integrative ideology has been applied to monitoring algal communities in order to assess the environmental effects of pollution (Wells et al., 2007, Guinda et al., 2008, Guinda et al., 2013). This type of integrative monitoring is an important toolset for advising on the conservation and strategic management of the marine environment.

There is potential for benthic EQRs to standardise benthic habitat monitoring outputs across the Directives (Borja, 2005). They provide a quantitative measure of environmental status, as required by each Directive. Using ES classifications in combination with standard biotope classifications is a holistic approach to describing the composition, health and functioning of the benthic environment.

1.3 Overcoming confounding variability in faunal assemblages

Benthic communities are naturally variable in time and space (Kennedy et al., 2011). Both reef and sedimentary communities have been independently described as random patchy ‘mosaics’, in which each patch contains the same community but at different stages of succession (Johnson, 1971, Johnson, 1972, Sousa, 1979a).

When assessing benthic community health in relation to point sources of disturbance, high background variability in faunal community structure can act as a confounding effect if the survey is not adequately designed or replicated (Underwood, 1991, Underwood, 1994, Underwood and Peterson, 1988a). This issue becomes more pertinent within poorly understood environments (Underwood, 1991, Underwood, 1994, Underwood and Chapman, 1996). In some instances robust experimental layouts, standard multivariate faunal, and univariate biodiversity index analyses do not suffice in detecting significant change in benthic community structure (Borja et al., 2003, Reiss and Kröncke, 2005, Kröncke and Reiss, 2010, Forde et al., 2015). Multimetric indices have proven to be robust against the effects of high background variability in community structure (Jeffrey et al., 1985, Schimmel et al., 1994, Deegan et al., 1997, Forde et al., 2012, Forde et al., 2015).

1. Introduction

Forde et al., (2015) assessed the impacts of Oyster (*Crassostrea gigas)* cultivation has on intertidal mudflats, which were designated as Natura 2000 sites. The authors found there to be a significant impact of sand compaction caused by the heavy vehicles that used access routes to the trestles. Standard multivariate and univariate diversity analyses could not detect the compaction impact across all sites. ES was assigned to each treatment area at each site using the IQI (Prior, 2004, Mackie, 2009, Phillips et al., 2014). This resulted in a generalised mixed linear model detecting a higher probability of ES classifications to fall below the "Moderate"/ "Good" boundary in access routes compared to other treatment areas. This result indicates that benthic habitat fragmentation could occur along mudflats that host trestle aquaculture activities. This finding was significant as mudflats that are not covered at low tide are a qualifying feature of interest under the HD. Mudflats are also important feeding grounds for native and migratory birds protected under the BD.

The multivariate and univariate diversity analyses used by Forde et al. (2015) were hindered by the variability in community structure. The trestle sites that were surveyed are flushed by tides twice daily, this coupled with the mobile nature of sandflat habitat creates a harsh physical environment for resident fauna. This was reflected in the low number of species, low univariate diversity index values and high proportion of first order opportunists found at each site.

Similar difficulties are likely to arise for studies carried out in other high energy, variable environments such as tidal rapids. Tidal rapids have the potential to provide significant amount of marine renewable energy through tidal energy extraction (Shields et al., 2008). Tidal rapids have been comparatively understudied compared to the sedimentary environment as a result of the difficult working conditions associated with them. These tidal rapids contain mostly reef substrate as sedimentation does not occur in areas with high current velocities (Warwick and Uncles, 1980, Conor et al., 2004). These reefs tend to be dominated by stress tolerant opportunistic epifauna (Savidge et al., 2014). Tidal rapids are highly dynamic and host unique and biodiverse benthic communities (Wilding et al., 2005; Savidge et al., 2014) and as a result are frequently designated as Natura 2000 sites.

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When a tidal stream turbine is operational a turbulent wake is created downstream of the device. This turbulent wake has the potential to negatively impact the benthic environment up to the point where the wake mixes back into the bulk flow, restoring homogenous conditions (Polagye, 2011). Within a turbulent wake, modified flow conditions may result in the scouring or removal of fauna due to accelerated flow around large bottom structures and substrates, or through the mobilisation of substrates (Daly and Mathieson, 1977, Palmer and Palmer, 1977, McGuinness, 1984, McGuinness, 1987b, McGuinness, 1987a, McGuinness and Underwood, 1986).

To definitively assess the potential impact of flow modification on epifaunal reef communities, it is essential to identify a relevant sensitivity measure (Andrew and Mapstone, 1988, Bishop et al., 2002). In traditional sedimentary benthic monitoring exercises, successional paradigms have been the basis for assessing whether or not a benthic community is in an unfavourable condition relative to control conditions (Pearson and Rosenberg, 1978, Borja et al., 2000). Within reef environments, bare rock is the successional start point, or totally disturbed condition (Chapman, 2002a), and it is from this point that the reef community develops from the first colonising propagules. If an EQR could be produced for use in subtidal animal dominated reefs it would improve the power of the studies to detect higher order changes in community structure that traditional monitoring techniques can miss (Forde et al., 2015, Savidge et al., 2014) . A multimetric index for use on reefs will also help to integrate the outputs from reef and sedimentary assessment studies in response to the Directives.

When monitoring benthic habitats it is important to be aware of natural long-term fluctuations in global meteorological conditions and short-term sporadic weather events which can affect ocean conditions in an unpredictable manner (Halpern et al., 2015, Kennedy et al., 2011). Seasonality has a relatively predictable effect on faunal abundances and this confounding effect can be avoided by targeting the same time period each year for sampling (Kennedy et al., 2011).

The standard framework used for reporting on the 'status' habitats in Natura 2000 sites is the European Nature Information System (EUNIS). The EUNIS habitat classification system is a hierarchical system that produces semiquantitative descriptions of the biological and physical characteristics of a habitat known as Biotopes. There are 6 levels of biotope, levels 1 - 4 describe with increasing detail, the depth zone, substrate type, exposure and salinity. Levels 5 and 6 describe the physical environment and the characterising fauna likely to be found there *sensu* Conor et al., (2004) .

The biological records used in the EUNIS classifications are not based on time averaged community structure per physical habitat, but on once off surveys carried out in the same habitat type in different areas. This has implications for long-term monitoring studies as the actual longevity of the biological communities reported to be present in each habitat is likely to vary due to inter-annual variability (Junker et al., 2012). This could potentially make it difficult to determine what the 'stable' state community within a habitat is over time. Effective implementation of the MSFD will require temporally stable baseline information for benthic habitats in coastal and offshore environments. It would be prudent to use level 4 biotope descriptions (Vasquez et al., 2015, Coltman et al., 2008) as long-term baseline precedents, as they contain good information on the physical environment of sedimentary biotopes, and are likely to persist over significant time periods (Gray, 1981).

There is potential for benthic EQRs to be used in applied impact assessments in the offshore environment (Borja et al., 2010b) under the MSFD (Barry et al., 2013). Such sources of disturbance are likely to occur around construction sites, aggregate (Desprez, 2000), mineral and hydrocarbon extractions sites (Stephens and Diesing, 2015) and intensively trawled areas (Engel and Kvitek, 1998, De Groot, 1984). Using EQRs to establish a baseline precedent prior to the commencement of construction or extraction processes provides information on the 'status' of a habitat. This allows for long term monitoring practices that are more robust against seasonal and interannual variability than standard data analyses (Reiss and Kröncke, 2005, Kröncke and Reiss, 2010).

1.4 Efficient benthic mapping techniques

One of the major issues facing EU member states with large EEZs is the large cost associated with mapping and monitoring the benthic environment. The bottom-up mapping approach is prohibitively costly at large spatial scales, yet the top-down procedure may require significant investment in hydrodynamic model development and large scale acoustic seabed classification surveys. It is important to understand the benefits and limitations of these methods. Proper understanding of the resources available to conservation initiatives allows for monetary allocations to be managed in a way that maximises the effectiveness of the exercise (Clements et al., 2010).

The key environmental drivers of benthic macrofaunal distributions vary with spatial scale (Andrew and Mapstone, 1988). In large areas that incorporate multiple physical habitat types, abiotic factors such as ocean currents structure benthic macrofaunal distributions through food provision (White et al., 2005, Mienis et al., 2007, Tweedle, 2005), the distribution of larvae (Cowen and Sponaugle, 2009) and through influencing the seafloor substrate characteristics (Sarkar, 2000, Dos Santos Brasil and Goncalves da Silva, 2000, Posey and Ambrose Jr, 1994). These processes are integral to ecosystem functioning and must be properly understood to allow for adequate conservation and management of ecosystems At local habitat scales the interaction of currents with benthic macrofauna are highly variable and dependent on the ecological resolution observed (Underwood and Chapman, 1996, Underwood and Petraitis, 1993, Wootton, 2001, Terlizzi et al., 2007, Vogel, 1994, Vogel, 1996, Okamura, 1984, Okamura, 1985, Okamura, 1988). High amounts of variability in animal-flow interactions at local scales may have implications for applied impact assessment studies assessing fine scale resolution effects of flow modification as a result of anthropogenic activities. Studies that have effectively used hydrodynamic parameters to explain spatial variation in benthic macrofaunal distributions have focussed on coarse spatial resolutions (Dutertre et al., 2013, Guinan et al., 2009a, Rengstorf et al., 2014). The ability of hydrodynamic models to explain variability in benthic community structure over large areas could make them useful tools for large scale mapping studies required to meet the criteria of the MSFD.

There is an increasing need for easily acquired and representative response (Dalleau et al., 2010) and predictor (Brown et al., 2011) variable surrogates in large scale mapping exercises. Adopting surrogates for traditionally used variables can significantly reduce costs and survey effort if data acquisition for the traditional variables is time consuming. High resolution sediment grain size data is frequently classified as coarse resolution nominal descriptors when reporting on the conservation status of sedimentary habitats. Studies carried out in designated Natura 2000 sites commonly use the EUNIS sediment classification system (Long, 2006) when reporting on the status of sedimentary habitats (Kennedy, 2008). Recent developments in acoustic mapping technologies present new time effective methods of seafloor data acquisition over large spatial scales (Brown et al., 2011). Acoustic signal derivatives have been successfully used as surrogates for sediment particle size analysis (PSA) outputs in large scale benthic habitat mapping studies (Brown et al., 2011, Brown and Blondel, 2009, Buhl-Mortensen et al., 2009, Callaway et al., 2009, Cook et al., 2008, Ehrhold et al., 2006, Dolan et al., 2008, Dolan et al., 2009). Issues such as fine scale sediment heterogeneity and gradational changes in sediment characteristics can affect acoustic derivative representativeness at small spatial scales (Brown and Collier, 2008). However, these issues become less pertinent with increasing spatial scale, or within areas of homogenous habitat (Monteys et al., 2016).

The ‘Irish National Seabed Survey’, now INFOMAR (Integrated Mapping for the Sustainable Development of Irelands Marine Resource), has produced acoustic sediment classifications for a significant proportion (125,000 km^2) of Ireland’s EEZ. This publically available data has the potential to be central to the development of spatial models that can predict the distributions of benthic communities as is required under the MSFD.

Using publically available data sources in conjunction with time-efficient survey techniques could significantly reduce costs associated with large scale surveys and subsequent laboratory analyses and report writing. Sub sampling sediment samples from single grabs has been shown to be an cost-effective survey technique (Somerfield and Clarke, 1997). The limitations of this technique are more pertinent when applied to coarse sediments, and when used in high resolution habitat quality assessments (Kennedy et al., 2011). This must be taken into consideration during survey planning stages. Cost effective survey designs are integral to maximising survey outputs within budgetary limits (Clements et al., 2010), however, it is important to ensure survey designs do not affect the integrity of the outputs (de Jonge et al., 2006). Expert judgement on sediment type in the field could potentially be an effective method of immediately classifying sediments into EUNIS categories. This is a reasonable expectation as other kinds of ecological quality assessments are based on methods carried out by experts in the field (Munné et al., 2003, Wells et al., 2007).

1.5 Thesis aims

It is the aim of this thesis to propose methodologies that could act as solutions for some of the challenges faced by benthic habitat assessments carried out in response to EU Directives.

Thesis aims:

- To test the flexibility of WFD benthic EQRs and their performance in totally new environments.
- To assess the effectiveness of hydrodynamic model outputs as predictor variables for benthic communities at small (1 m^2 grid size) and medium (100 m) scales.
- To test if cost effective survey techniques can be used in habitat mapping studies aimed at conserving the benthic environment.
- To assess how designated waterbodies around Ireland are being effected by anthropogenic impacts using aquaculture and tidal energy extraction as potential stressors.

2. Case studies

Here, the synopses of six manuscripts are presented individually, followed by a general discussion addressing each of the thesis aims. Each case study is present in full manuscript form in the Article Chapters preceding this thesis. Only key results are presented here for the purpose of clarity. The reader is referred to the relevant Article Chapter and section for in-detail methods and results.

2.1 Case study 1

Author declaration: My contribution to this manuscript involved; field work, all taxonomic analyses, supervision of particle size analysis (PSA).

This study was published in the Marine Pollution Bulletin Journal on the 15th of June 2015:

Impact of intertidal oyster trestle cultivation on the Ecological Status of benthic habitats

James Forde, Francis X. O'Beirn, Jack P.J. O'Carroll, Adrian Patterson, Robert Kennedy

Abstract

A considerable number of Ireland's shellfish production areas co-occur with or are adjacent to Natura 2000 sites which are protected under European legislation. To investigate the general interaction between trestle oyster cultivation and the surrounding intertidal environment, six sites were selected within designated Natura 2000 sites. At each trestle site three Treatment areas were sampled. One Treatment area corresponded to potential impacts associated with cultivation activities occurring at trestle structures (designated the Trestle Treatment) while one Treatment area corresponded to potential impacts due to cultivation activities occurring along access routes (the Access Treatment). An area not subject to any known anthropogenic activity was used as a control (the Control Treatment). Potential impacts associated with Trestle Treatment areas included changes in sediment total organic matter (TOM) levels underneath trestles due to the bio-deposition of faecal/pseudofaecal material while the predominant impact associated with Access Treatment areas was compaction of sediments due to heavy vehicle traffic. In this study, macrobenthic communities at the sites were highly variable and exhibited low levels of diversity which prevented the detection of general effects of cultivation activity on community

structure, diversity and secondary production. To overcome this variability, the Infaunal Quality Index (IQI) was used to assess impacts on ES of benthic communities (WFD). Relative to Control and Trestle Treatment areas, activities occurring at Access Treatment areas had a significant negative impact on ES. This study highlights the potential of the IQI for the management of aquaculture activity and provides validation for the use of the IQI in Irish intertidal environments. This study also highlights the IQI as a potential tool for assessing the conservation status of designated habitats in Natura 2000 sites.

For a detailed description of this study refer to the manuscript in Article Chapter I.

2.1.1 Case study 1 synopsis

Given the frequent occurrence of oyster trestle cultivation within Natura 2000 sites an appropriate assessment was carried out to assess the associated impacts of trestle cultivation. Existing literature on the impacts of trestle cultivation on intertidal benthic communities are inconsistent in regard to the type and extent of impacts that are reported. This study is aimed at definitively assessing the potential impacts of oyster trestle cultivation on intertidal benthic habitats at six cultivation sites that occur within Natura 2000 sites around Ireland.

Another aim of this study was to test the applicability of the WFD IQI EQR for the assessment of the putative impacts of shellfish cultivation on intertidal habitats and associated communities. Using the IQI would allow the potential effects of trestle cultivation on the ES of intertidal communities to be comparable across sites from different geographic locations. This will inform the further development of a systematic approach to measure and assess ES in intertidal areas in relation to anthropogenic activities.

The embayments/harbours sampled included Donegal Bay on the northwest coast (two sites), Clew Bay (two sites) on the west coast, and Dungarvan Harbour (Outer) (one site) and Bannow Bay (one site) on the southeast coast (Figure 2).

Three treatment areas were chosen at each site. One immediately beneath the trestles (Tr), one along the vehicle access route to the trestles (As) and the other 300 m away from any cultivation activities (Cl). Within each treatment area, quadrats measuring 50 m long and 3 m wide were identified. In all cases the positions of the sampling quadrats were accurately recorded using a handheld

GPS. Within each sampling quadrat two 0.01 m^2 cores were taken at ten randomly positioned stations, one for faunal analysis the other for sediment analysis.

At each station in-situ observations of sediment characteristics (e.g. sediment type/sorting, sediment REDOX layer depth, evidence of local burrowing activity) were recorded. At each station two 0.01m^2 cores were retrieved, one for faunal analysis and one for sediment analysis. Faunal core samples were sieved through a 500 µm mesh on site. The material remaining on the sieve was fixed in 4% formalin solution prior to laboratory analysis. All fauna were identified to species when possible during taxonomic analysis. All sediment granulometry and organic carbon analyses were carried in accordance with the NMBAQC guidelines (NMBAQC, 2009). Following identification, ash-free dry weight (AFDW) biomass for each taxa within the replicate faunal core samples was determined using the method outlined in Rumohr (1990, 2009).

In the current study, total community secondary production estimates were derived for each replicate faunal core using the freely-available empirical secondary developed by Brey (2001, http://www.thomasbrey.de/science/virtualhandbook/navlog/index).

Sediment granulometry was determined for each pre-treated subsample using a combination of laser particle sizing (LPS) and wet/dry sieving (Eleftheriou and McIntyre, 2005; Forde et al., 2012). All sediment samples were classified as Fine Sand (Folk and Ward, 1957). Mz is a parameter used to describe the mean particle size of a distribution and is analogous to the graphic mean employed with the normal distribution in conventional statistics (Forde et al., 2012).

For each dried sediment sample, organic content was determined by weight loss on combustion of 5 g of sediment at 450 °C after 6 h (Eleftheriou and McIntyre, 2005).

Analyses were carried out using PRIMER v6 (Clarke and Gorley, 2006a)with the add-on package PERMANOVA+ (Anderson et al., 2008), MINITAB v16 and R. Using untransformed faunal data the DIVERSE routine in PRIMER v6 (Clarke and Gorley, 2006a, Clarke et al., 2006) was used to calculate a range of diversity measures for each replicate faunal core. Diversity measures calculated included the total number of taxa (S), total number of individuals (N), Shannon diversity index (H’

Loge) and Simpson's evenness diversity index (1—k'). SIMPER analysis (Clarke, 1993a) was carried out in square root transformed faunal abundance data to identify the characterising species for each treatment area. The Treatment areas across the six sampling sites conformed to the EUNIS level 4 Biotope polychaete/bivalve-dominated muddy sand shores, a common biotope often found as extensive intertidal flats on open coasts and in marine inlets (Conor et al., 2004). In general, infaunal communities associated with the Treatment areas comprised a range of polychaetes, bivalves, amphipods and gastropods.

A global PERMANOVA was carried out to test for the effect of Treatment on diversity measures, multivariate community structure, biomass, standardised TOM, and binary ES responses along the Moderate/ Good boundary on any of the aforementioned response variables. However, results did indicate significant differences in community structure, biomass, diversity measures and secondary production values across the six sites (Site (Treatment) P-value < 0.05). This indicates that the six sites had different identities in terms of faunal community composition and community secondary production. Similarly, Treatment had no significant effect on standardised TOM, whereas Treatment nested in Site did significantly effect standardized TOM (Site (Treatment) P- value < 0.001).

The frequency distribution of cores classified as High/Good or Moderate/Poor/Bad within each Treatment is presented in Table 2. Results of a generalised linear mixed model with a binomial distribution and a logit link function, (run using the glmmML package in R 3.0.2 (R Development Core Team, 2014)) indicated a significant difference between the Access Treatment relative to the Control and Trestle Treatments in terms of the proportion of cores occurring within the High/Good ES and the Moderate/Poor/Bad ES categories (P-value = 0.033. See Table 3). Relative to both the Control and Trestle Treatments the Access Treatment had a significantly lower proportion of cores classified as High/Good and a significantly higher proportion of cores classified as Moderate/Poor/Bad.

2. Case studies

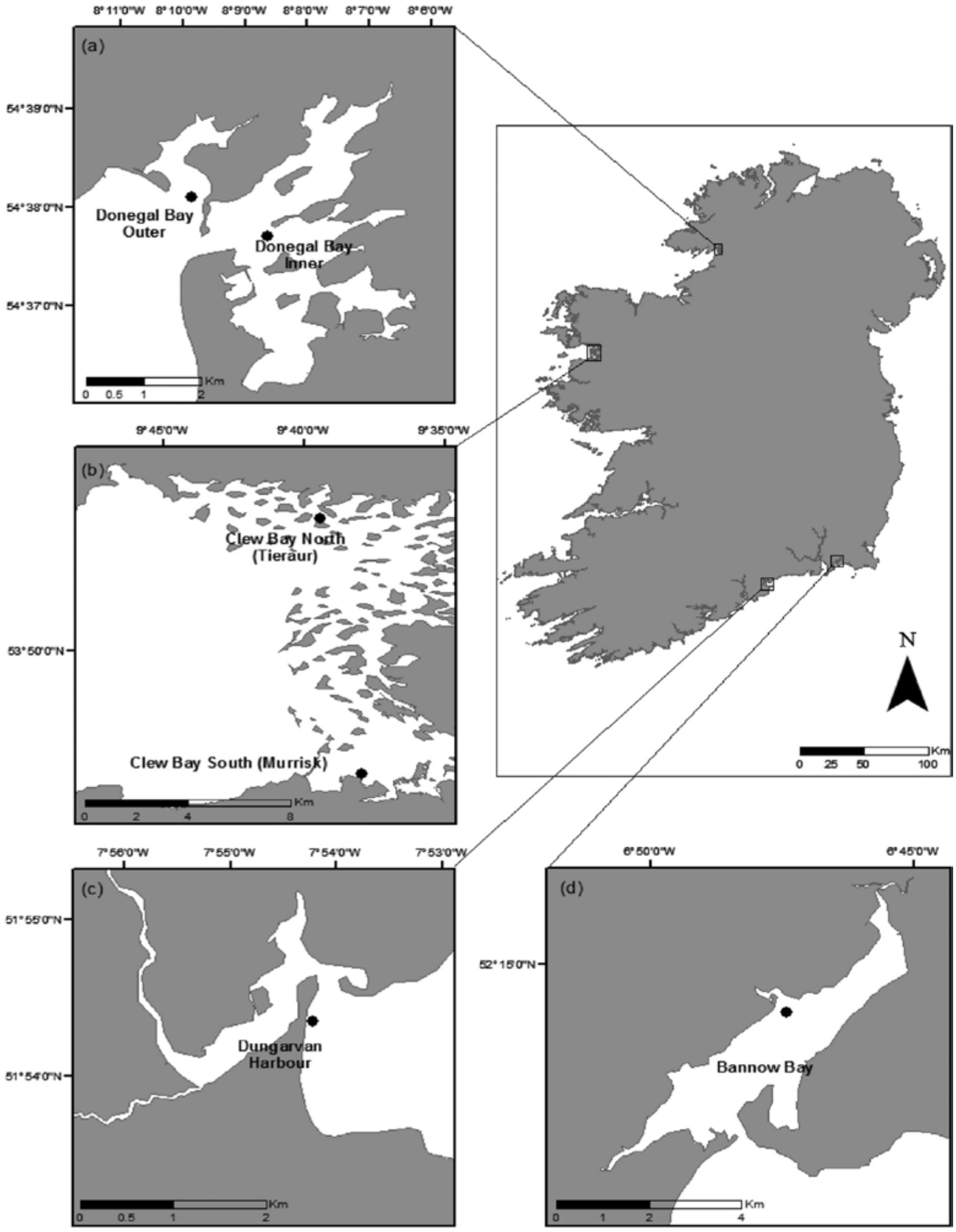

Figure 2: General location of intertidal oyster trestle cultivation sites sampled in October/November 2013 to investigate the potential impacts of cultivation activities on habitat sediment characteristics, macrobenthic community structure, diversity, secondary production and Ecological Status. All cultivation sites occur within designated Natura 2000 sites.

A global PERMANOVA analysis was carried out to test for the effect of Treatment on diversity measures, multivariate community structure, biomass, standardised TOM, and binary ES responses along the Moderate/Good boundary. There was no significant effect of treatment.

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IQI ES category	Trestle	Access	Control
High/Good	45	32	49
Moderate/Poor	15	28	11

Table 2: ES classification of faunal cores at oyster trestle cultivation Treatment areas (i.e. Trestle, Access, Control). ES determined using the Infaunal Quality Index (IQI).

Source	**Coeef**	**SE(Coef)**	**z**	**Pr (>| z|)**
Intercept (Control)	2.379	1.5924	**1.494**	**0.1350**
Factor (Access Treatment)	-1.6224	0.5522	**-2.938**	**0.0033**
Factor (Trestle Treatment)	0.5537	0.5309	**1.043**	**0.2970**
Residual: 151.3	**df**: 176	**AIC**: 159.3.		

Table 3: A generalised linear mixed model (run in the glmmML package in R 3.0.2) used to investigate the association between oyster trestle cultivation activity Treatment (i.e. Trestle, Access, Control) and ES classification of faunal cores. ES determined using the Infaunal Quality Index. The glmmML is a generalised model based on a binomial distribution using and a logit link function. Null hypothesis; no significant difference in the probability of a core having High/Good ES at different levels of the factor Treatment. *P*-values obtained from a likelihood ratio (LR) test. Significant values ($P < 0.05$) in **bold**.

The significantly lower ES along access routes is caused by the compaction of sediments as a result of heavy vehicle traffic crossing the sandflats to trestle cultivation areas. This effect may only be a seasonable phenomenon that is removed by the resuspension of sediments by winter storm events. Further research is needed to assess the longevity of this impact.

2.2 Case study 2

Author declaration: My contribution to this study included; field work, taxonomic analyses (50%), supervision of sediment PSA, data processing, a significant majority of statistical analyses, and the writing of the manuscript.

This study was published in the Marine Pollution Bulletin Journal on the 15th of September 2016:

Impact of prolonged storm activity on the Ecological Status of intertidal benthic habitats within oyster (Crassostrea gigas) trestle cultivation sites

Jack O'Carroll, Christina Quinn, James Forde, Adrian Patterson, Francis X. O'Beirn, Robert Kennedy

Abstract

The ES of intertidal benthic communities within six oyster trestle cultivation sites was found to be negatively impacted along the access routes to trestles in a 2013 study. All cultivation sites occur within Natura 2000 sites.

The current study revisited four of the 2013 cultivation sites in February 2014 one month after the storm activity of winter 2013/14 to test if the compaction effect along access routes persisted after the storms.

Three levels of the fixed factor treatment were sampled; immediately below the trestles, along the access route and 300 m away from any anthropogenic activity.

The compaction effect at the Access treatment persisted in spite of the major storm activity. The current study showed the IQI to be effective for assessing the impacts of aquaculture and highlights the IQI as a tool for monitoring Conservation Status of intertidal communities under the Habitats Directive.

For a detailed description of this study refer to the manuscript in Article Chapter II.

2.2.1 Case study 2 synopsis

This study is a follow up study to that of Forde et al. (2015), who sampled six oyster trestle cultivation sites around Ireland in November 2013, and found that the access route impact was present across the six sites. The compaction effect is caused by compacting of sediment by heavy vehicles crossing the sandflats to the trestle cultivation areas.

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This study revisited four of the six sites (See Fig. 3) in February 2014 immediately after the significant stormy period that impacted coasts in Ireland and the UK (http://www.met.ie/climate-ireland/weather-events/winterstorms13_14.pdf).
The aim of this study was to test if the compaction effect persisted after the significant resuspension of sediments that occurred during the natural disturbance events of winter 2013, 2014. The premise being that if the access impact can withstand significant natural disturbance events, it may be a permanent intertidal habitat fragmentation effect. The IQI was used to monitor trestle activities in this study and in Forde et al, (2015). The results of this study give further useful information on its application in a new, highly variable environment.

Three Treatment areas were chosen at each site. One immediately beneath the trestles (Tr), one along the vehicle access route to the trestles (As) and the other 300 m away from any cultivation activities (Cl). Within each treatment area, quadrats measuring 50 m long and 3 m wide were identified. In all cases the positions of the sampling quadrats were accurately recorded using a handheld GPS. Within each sampling quadrat two 0.01 m^2 cores were taken at ten randomly positioned stations, one for faunal analysis the other for sediment analysis. At each station in-situ observations of sediment characteristics (e.g. sediment type/ sorting, sediment REDOX layer depth, evidence of local burrowing activity) were made. Faunal core samples were sieved through a 500 µm mesh on site. The material remaining on the sieve was fixed in 4% formalin solution prior to laboratory analysis. All sediment granulometry and organic carbon analyses were carried in accordance with the NMBAQC guidelines (NMBAQC, 2009).

All of the data gathered for this study was compared to that of Forde et al. (2015) to allow for before and after comparisons relative to the winter storm events to be made. A global fixed effects model was built using PERMANOVA (Anderson, 2008) to test for the effect of time (Ti: two levels; before and after storms), site (Si: four levels; Dungarvan, Bannow, Donegal Inner and Donegal Outer) and Treatment (Treat: three levels: Tr, As, Cl) on univariate diversity index outputs, square root transformed faunal abundances, IQI outputs and sediment organic content.

2. Case studies

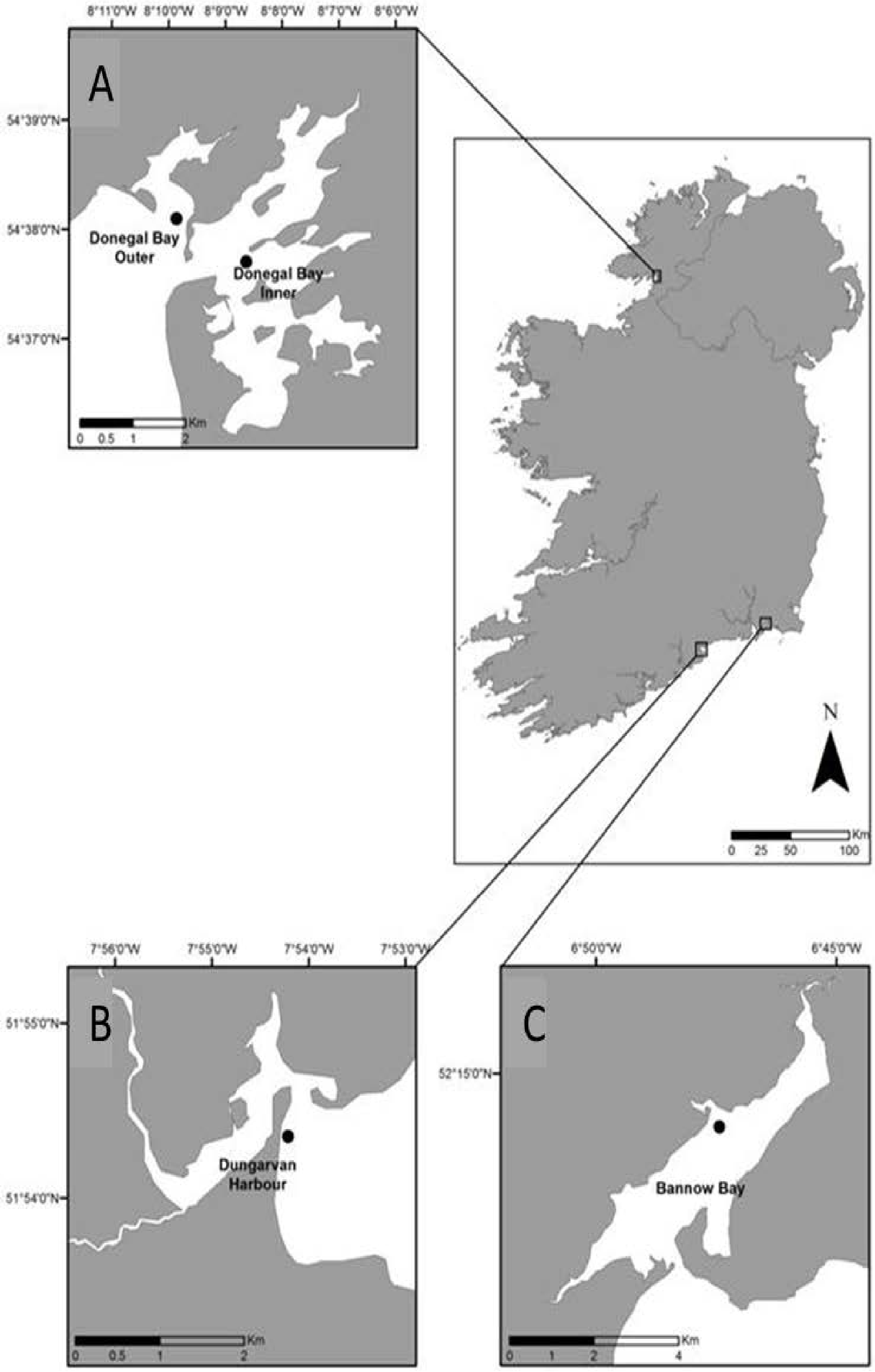

Figure 3: Map of oyster cultivation sites sampled in October-November 2013 and February 2014.

2. Case studies

Sediment grain size distributions were largely similar at each site. There was a significant reduction in mean sediment grain size and organic content over time due to sediment resuspension.

The biotope description of each site remained the same over time and conformed closely to the level 4 biotope; polychaete/bivalve- dominated muddy sand shores. Univariate diversity measures were calculated for each site using the DIVERSE routine in PRIMER v6 (Clarke and Gorley, 2006a) (No. of taxa; S. Total number of individuals; N. Shannon diversity index; H. Simpson's evenness diversity index; 1-ʎ'). There was a significant effect of the random factor Site, indicating the geographic variation between the sites. The diversity measures at each site were unaffected by treatment. There was an almost twofold increase in the number of individuals from 2013 to 2014, this was caused by an increase in the number of small first order opportunistic taxa such as the polychaete *Pygospio elegans.* However, this was not found to be significant by PERMANOVA.

Square root transformed multivariate faunal abundances at each site were significantly different and changed significantly over time, but were unaffected by treatment. Continuous EQR values produced by IQI did not significantly change with time, were unaffected by treatment and were significantly different at each site (See Table 4).

Source	**df**	**SS**	**MS**	**Pseudo-F**	**P(perm)**	**Unique perms**	**P(MC)**
Treat	2	0.155	0.077	0.259	0.758	997	0.772
Ti	1	0.225	0.225	1.358	0.265	996	0.261
Si(Treat)	9	2.691	0.299	25.081	0.001	999	0.001
TrxTi	2	0.029	0.014	0.088	0.902	997	0.913
TixSi(Treat)	9	1.495	0.166	13.93	0.001	999	0.001
Res	214	2.551	0.011				
Total	237	7.128					

Table 4: PERMANOVA table of results for IQI continuous EQR values.

A generalised linear mixed model with a binomial distribution and a logit link function was run using the glmmML package in R 3.0.2 (R Development Core Team, 2014) . The response variable used is a binary distribution of IQI ES classifications along the critical 'Moderate/Good' boundary (See Table 5). Over time the access route differed significantly to the control treatment in terms of the distribution of the binary ES response variable around the 'Moderate/Good' boundary (p= 0.0298). The trestle treatment area was not significantly different to the control (p=0.159). The major storm event in the winter of 2013/14 had a significant negative impact on ES across all sites and treatment levels (p=0.0151). 58% of samples were classified as 'Good' or higher in 2013 compared to 40% in 2014 (Table 6).

Factors	**coef**	**se(coef)**	**z**	**Pr(>|z|)**
Intercept (Control area 2013)	0.7882	0.8288	0.9511	0.342
factor(Treat)TR	0.8111	0.5763	1.4074	0.159
factor(Treat)AS	-1.2391	0.5704	-2.1722	0.0298
factor(Time)2014	-1.3958	0.5743	-2.4305	0.0151
factor(Treat)TR:factor(Time)2014	-0.1884	0.8012	-0.2352	0.814
factor(Treat)AS:factor(Time)2014	1.0812	0.799	1.3532	0.176

Table 5: This table shows the results of a generalised linear mixed model with a binomial distribution and a logit link function. The response in this model was the distribution of HG ES classifications. This analysis was carried out to test for significant differences in the distribution of HG between the control (Intercept) treatment and the Access (AS) and trestle treatments (TR) over time (Yr) with the random factor of site removed. Access (AS) was significantly different to the control treatments across all sites over time ((Treat)(AS); p= 0.0151)). 2014 was significantly different to the intercept (2013) in terms of the distribution of ES classification across all treatments.

	Oct-13			Feb-14		
IQI ES category	Trestle	Access	Control	Trestle	Access	Control
HG	30	17	25	20	15	16
MPB	10	23	15	20	25	24

Table 6: This table shows the distribution of IQI ES classifications around the critical Moderate/ Good Boundary for each level of the random factor treatment across all sites in October 2013 and February 2014.

The persistence of the compaction effect along access routes in spite of significant disturbance events during the winter of 2013, 2014 indicates that it may be a permanent impact. Further research is needed to assess the best procedure to manage and possibly mitigate the risk this effect poses to the conservation status of Natura 2000 areas.

2.3 Case study 3

Author declaration: My contribution to this manuscript involved: Developing the proposed EQR, revisiting this data and applying all new statistical analyses and writing the manuscript.

This study has been accepted for publication at Ecological Indicators and will be published in February 2017:

Identifying Relevant Scales of Variability for Monitoring Epifaunal Reef Communities at a Tidal Energy Extraction Site

J.P.J. O'Carroll, R. M. Kennedy, G. Savidge

Abstract

The SeaGen tidal energy turbine is located in the Strangford Narrows, Northern Ireland. The Narrows are designated as a Natura 2000 site, host unique biological assemblages and exhibit very high tidal velocities.

This study describes an asymmetrical BACI design monitoring program that was aimed at assessing the potential impact the SeaGen may have on epifaunal boulder reef communities. This study presents a novel methodology for monitoring epifaunal communities within highly variable and poorly understood tidal rapid environments.

We identify bare rock as a key measure of disturbance within tidal energy extraction sites and propose a new successional model for epifaunal reef communities on subtidal stable substrates. We also present an Ecological Quality Ratio (EQR); the High Energy Hard Substrate (HEHS) index for use in monitoring programs within tidal energy extraction sites.

The SeaGen is not having a significant negative effect on epifaunal communities, bare rock distributions or EQR values at the impact site. Seasonality significantly affected all stations equally over time. The HEHS index detected patches of disturbance but this fell within the range of natural variability for the habitat. The HEHS index has the potential to standardise benthic monitoring in tidal energy extraction sites.

For a detailed description of this study refer to the manuscript in Article Chapter III.

2.3.1 Case study 3 synopsis

This study describes a before after control impact (BACI) study with the aim of presenting a new methodology for monitoring the effects of tidal energy extraction on epifaunal reef communities. The study area lies within the Strangford Lough Narrows, Northern Ireland, which is designated under the HD and BD. Tests for differences between the control stations and the impact station in terms of multivariate faunal distributions were carried out, a standard method for this kind of study. Bare rock was also assessed in statistical analyses and the proportion of bare rock is suggested as a relevant measure of physical disturbance for epifaunal reef communities. A successional model is proposed that describes how epifaunal communities change along a gradient of physical disturbance caused by events that are a result of hydrographic variation (See Figure 4). Bare rock distributions and the proposed successional model are incorporated into a WFD style multimetric index; the High Energy Hard Substrate Index (See Equation 1). The HEHS index is a useful and effective tool for benthic monitoring within tidal energy extraction sites that host animal dominated, sublittoral reefs.

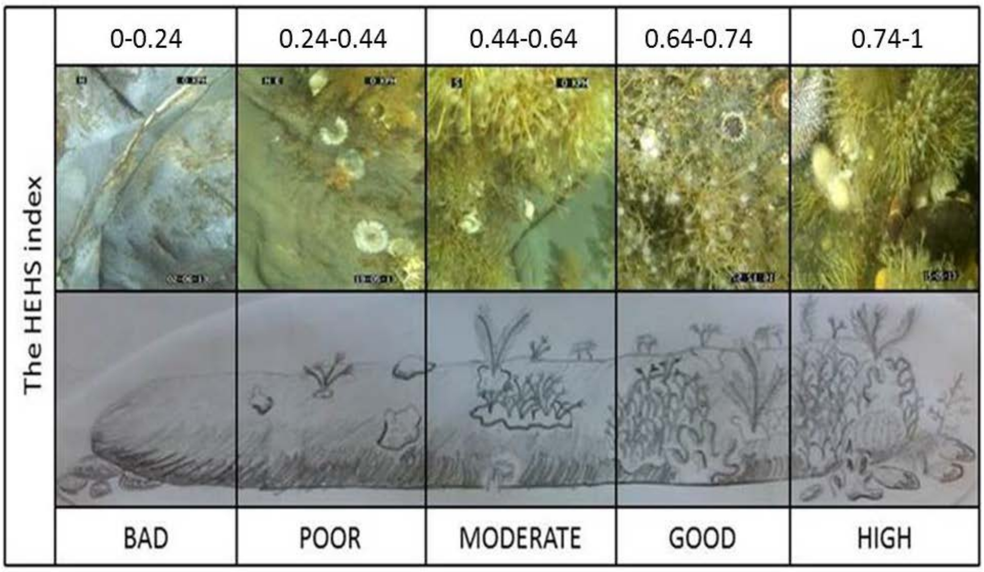

Figure 4: This figure depicts the successional paradigm on which the HEHS is based. The successional change gradient is divided into five parts by class boundary cut-off points at four intervals between 0 and 1. Bad=0-0.24, Poor=0.24-0.44, Moderate=0.44-0.64, Good=0.64-0.74, High=0.74-1

Stations were sampled at 20 m (Impact station), 150 m (Control Station 1) and 300 m (Control Station 2) south-southeast of the turbine along the axis of the Narrows, with one station located 50 m (Control Station 3) east of the turbine

(See Figure 5). In this study an asymmetrical distribution of control versus impact stations was used in a BACI design experiment *sensu* Underwood (1991, 94).

The HEHS index

HEHS=

$$\frac{\left(-\left(\frac{Pi\ BR}{Max\ BR}\right)\times 0.3\right)+\left(\left(\left(\frac{H'}{H'\ max}\right)\times 0.1\right)\right)+\left(\left(\left(\frac{S}{S\ max}\right)\times 0.1\right)\right)+\left(\left(\frac{\left(\left(\frac{\%\ massive}{Max\ obsv.\%\ masv.cover}\right)+1\right)}{\left(\left(\frac{\%\ crust}{Max\ obsv.\%\ crust\ \ cover}\right)+1\right)}\right)\times 0.3\right)}{0.5}$$

(Equation 1)

Where; *Pi BR* = Decimal proportion of bare rock
H' = Shannon Wiener diversity index
S = Number of species
% massive = Percentage coverage of massive taxa
% crust = Percentage coverage of encrusting taxa
max = The maximum value for the metric from reference conditions.

At each station, five adjacent video quadrats were sampled using a 0.5 m x 0.5 m quadrat divided into 25 10 cm x 10 cm cells. Each cell was filmed in close up using a digital video camera. The pre-installation samples were taken in March 2008. Follow-up surveys took place in July 2008, March 2009, July 2009, April 2010 and April 2011. The water depth was 25-27m at all stations. The stations were precisely relocated using USBL acoustic marking devices attached to weighted marker frames on the seafloor. There were two occasions when 4 rather than 5 quadrats were sampled at a station because of the very difficult working conditions on the seafloor at the Narrows. Percentage cover estimation for fauna was carried out via video still analysis. Substrate distributions and the percentage bare rock were also derived from video still analysis.

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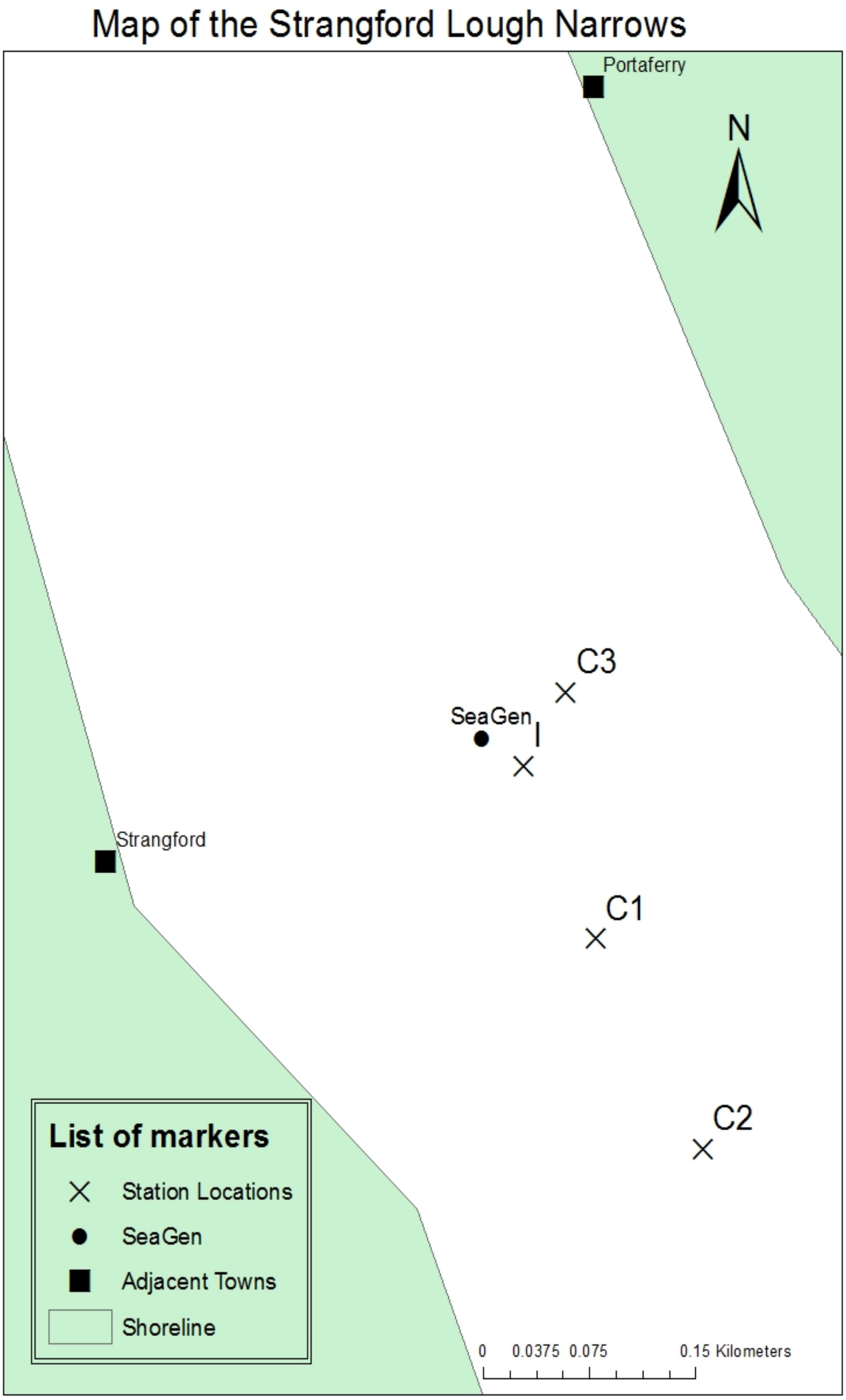

Figure 5: Location of the SeaGen tidal stream turbine, the position of the impact station (I) and the control stations (C1, C2, C3) within the Strangford Lough Narrows.

A mixed model PERMANOVA (Anderson, 2008) was used to test for differences between the impact and control stations in terms of square root transformed epifaunal abundances, bare rock distributions and continuous EQR value outputs produced by the proposed HEHS index. The fixed factor before/after (BA) had two levels: before (B) and after (A). The fixed factor control/impact (CI) had two levels control (C) and impact (I). The random factor Season (Se) was nested within BA, while the random factor Station (Stn) was nested within CI.

Epifaunal communities at each station were largely characterized by similar species over time with the hydroids *Sertularia argentia* and *Tubularia indivisa* being the most abundant at each station. The relative abundances of the other taxa varied considerably more over time.

Multivariate community structure did not change at the impact station in a way that differed to any of the controls over time. Community structure at all sites was significantly affected by seasonality (See Table 7). This is due to the seasonal settlement of opportunistic bryozoans and tubiculous amphipods during the summer sampling times of 2008 and 2009. Bare rock distributions were only significantly affected by seasonal change, similar to multivariate community structure. This indicates that relative faunal abundances and total faunal coverage are not being impacted by flow modification at the impact station.

EQR values were not affected by any of the factors in the mixed model. This result indicates that the HEHS index exhibits robustness against seasonality in a dataset, a trait shared by other well established multimetric indices (Kröncke and Reiss, 2010, Reiss and Kröncke, 2005). The HEHS index found 88 % of quadrats to have an ES of Good or High. The impact station was not significantly affected in a manner that differed to the other stations. The most spatially distant control station (C2) contained higher amounts of bare rock at the 3rd, 5th, and 6th sampling times. The HEHS index found this station to fall below the critical 'Moderate/Good' boundary at these times (See Table 8).

PERMANOVA table of results

Source	df	SS	MS	Pseudo-F	P(perm)	Unique perms	P(MC)
CI	1	1901.6	1901.6	0.94597	0.533	999	0.511
BA	1	15326	15326	1.3578	0.397	999	0.339
Stn(CI)	2	5029	2514.5	1.4341	0.146	998	0.188
Se(BA)	1	17593	17593	7.6282	0.041	996	0.001
CIxBA	1	1007.6	1007.6	0.61357	0.713	999	0.832
CIxSe(BA)	1	1497.9	1497.9	0.64949	0.539	998	0.677
BAxStn(CI)	2	5694.7	2847.3	1.6814	0.091	999	0.114
Stn(CI)xSe(BA)	2	4661.4	2330.7	4.2287	0.001	998	0.002
Res	58	62740	1081.7				
Total	69	1.15E+04					

Table 7: This table shows the results of the PERMANOVA on multivariate epifaunal community structure. The analysis was carried out on all levels of Time (Spring and Summer) and Station (I, C1, C2, C3). There was a significant effect of season nested in Before and After; Se(BA) P(perm)=0.041. There was also a random interaction effect of; Stn(CI)xSe(BA) P(perm) <0.001.

This anomalous occurrence is attributed to natural variability. The HEHS index appears capable of accounting for significant change of the kind that would be expected to persist in an area being negatively impacted by flow modification as a result of tidal energy extraction (for examples of EQR value breakdowns, see Table 8).

To definitively assess the potential localised effects of tidal energy extraction the relationship between epifauna and hydrodynamic conditions must be further elucidated. This will allow inferences to be made on the potential negative effects of modified flows in the wake of operational turbines.

Class Boundary	Max, Avg, Min EQR	Replicate	ES	EQR	Bare rock	M:C	H'	S
HIGH	MAX EQR	4_C1_2	H	0.995	0.024	3.283	1.338	11
HIGH	AVG EQR	2_I_5	H	0.867	0	0.867	1.919	19
HIGH	MIN EQR	3_C3 _5	H	0.755	0.112	1.343	1.304	13
GOOD	MAX EQR	5_C1_2	G	0.743	0.116	1.925	1.083	8
GOOD	AVG EQR	3_C2_1	G	0.69	0.361	5.204	1.134	12
GOOD	MIN EQR	5_C2_4	G	0.647	0.345	2.364	1.503	8
MODERATE	MAX EQR	5_C3 _5	M	0.634	0.016	0.596	1.272	7
MODERATE	AVG EQR	2_I_4	M	0.594	0.217	0.697	1.579	12
MODERATE	MIN EQR	6_C2_2	M	0.527	0.468	8.07	0.761	5
POOR	MAX EQR	4_C2_2	P	0.333	0.429	0.235	1.398	9
POOR	AVG EQR	-	-	-	-	-	-	-
POOR	MIN EQR	6_C2_4	P	0.272	0.788	8.489	1.106	5
BAD	MAX EQR	6_C2_5	B	0.191	0.826	1.761	1.22	6
BAD	AVG EQR	"	"	"	"	"	"	"
BAD	MIN EQR	"	"	"	"	"	"	"

Table 8: This table shows sample replicates that represent the maximum (MAX EQR), minimum (MIN EQR) and the real value closest to the mean EQR (AVG EQR), for each class boundary (H, G, M, P, B). Replicate= sample replicate label. ES= Ecological Status. EQR= EQR values for each replicate. Bare Rock= the Proportion of bare rock present. M: C= the Massive: Crust ratio. H'= the Shannon Weinner diversity Index. S= No. of Species.

2.4 Case study 4

Author declaration: My contribution to this manuscript involved: Data handling, some statistical analyses and helping with the writing of technical parts of the manuscript.

This study was published in PLOS one on the 25th of August 2016:

Do Changes in Current Flow as a Result of Arrays of Tidal Turbines Have an Effect on Benthic Communities?

L. Kregting, B. Elsäßer, R. Kennedy, D. Smyth, J. O'Carroll, G. Savidge[2]

Abstract

Arrays of tidal energy converters have the potential to provide clean renewable energy for future generations. Benthic communities may, however, be affected by changes in current speeds resulting from arrays of tidal converters located in areas characterised by strong currents. Current speed, together with bottom type and depth, strongly influence benthic community distributions; however the interaction of these factors in controlling benthic dynamics in high energy environments is poorly understood. The Strangford Lough Narrows, the location of SeaGen, the world's first single full-scale, grid-compliant tidal energy extractor, is characterised by spatially heterogeneous high current flows. A hydrodynamic model was used to select a range of benthic community study sites that had median flow velocities between 1.5–2.4 m/s in a depth range of 25–30 m. 25 sites were sampled for macrobenthic community structure using drop down video survey to test the sensitivity of the distribution of benthic communities to changes in the flow field. A diverse range of species were recorded which were consistent with those for high current flow environments and corresponding to very tide-swept faunal communities in the EUNIS classification. However, over the velocity range investigated, no changes in benthic communities were observed. This suggested that the high physical disturbance associated with the high current flows in the Strangford Narrows reflected the opportunistic nature of the benthic species present with individuals being continuously and randomly affected by turbulent forces and physical damage. It is concluded that during operation, the removal of energy by marine tidal energy arrays in the far-field is unlikely to have a significant effect on benthic communities in high flow environments. The results

are of major significance to developers and regulators in the tidal energy industry when considering the environmental impacts for site licences.

For a detailed description of this study refer to the manuscript in Article Chapter IV.

2.4.1 Case study 4 synopsis

The aim of this study was to investigate the interactions between epifaunal communities and a range of tidal velocities relevant to tidal energy extraction. Previous investigations have shown tidal energy extraction to have no significant effect on epifaunal communities 20 m downstream of a device (O'Carroll et al., 2017a). It remains possible that flow modification due to rotor action or large bottom mooring structures could have a complex or highly localised impact on ambient epifaunal communities. Assessing variation in epifaunal community structure in response to current velocities that are typically required for turbines to produce rated (maximum energy output) energy, will give an insight to the potential effects of flow modification caused by tidal energy extraction.

The study area exists within the Strangford Lough Narrows at least one kilometre away from the operational SeaGen tidal energy turbine to avoid any potential confounding influence of the tidal energy extraction process on this study.

The tidal regime of the Narrows was modelled using MIKE 21 modelling software (www.dhisoftware.com). Hydrodynamic model output cells were used to identify sampling sites that occur along a current velocity gradient with mean values from 1.5 ms^{-1} to 2.4 ms^{-1}. A drop down video camera was used to survey the seafloor at each site. Three replicate sites for each velocity class (1.5 ms^{-1}, 1.6 ms^{-1}, 1.7 ms^{-1}, 1.8 ms^{-1}, 1.9 ms^{-1}, 2 ms^{-1}, 2.1 ms^{-1}, 2.2 ms^{-1}, 2.4 ms^{-1}) were surveyed. At each site the camera system was allowed to settle for five to ten seconds at thirty different positions. This produced thirty random quadrats approximately two metres apart within a 20 m^2 area. The best twenty-one video stills out of the thirty taken at each replicate site were used for video still analyses. It was not possible to find a site with a velocity of 2.3 ms^{-1} using hydrodynamic model output cells, and it was only possible to produce one replicate for the 2.4 ms^{-1} velocity class. In total, twenty five sites were used in analyses (See Figure 6).

Percentage epifaunal coverage was converted into the SACFOR abundance scale *sensu* Conor et al., (2004). A Bray-Curtis similarity matrix (Bray and Curtis, 1957) was formed using the SACFOR transformed faunal data. SIMPER analysis was used to identify the characterising fauna at each site (See Table 9). Epifaunal communities were assigned to EUNIS biotopes. All sites were found to conform closely to the EUNIS level four biotope CR.HCR.FaT; very tide-swept faunal communities.

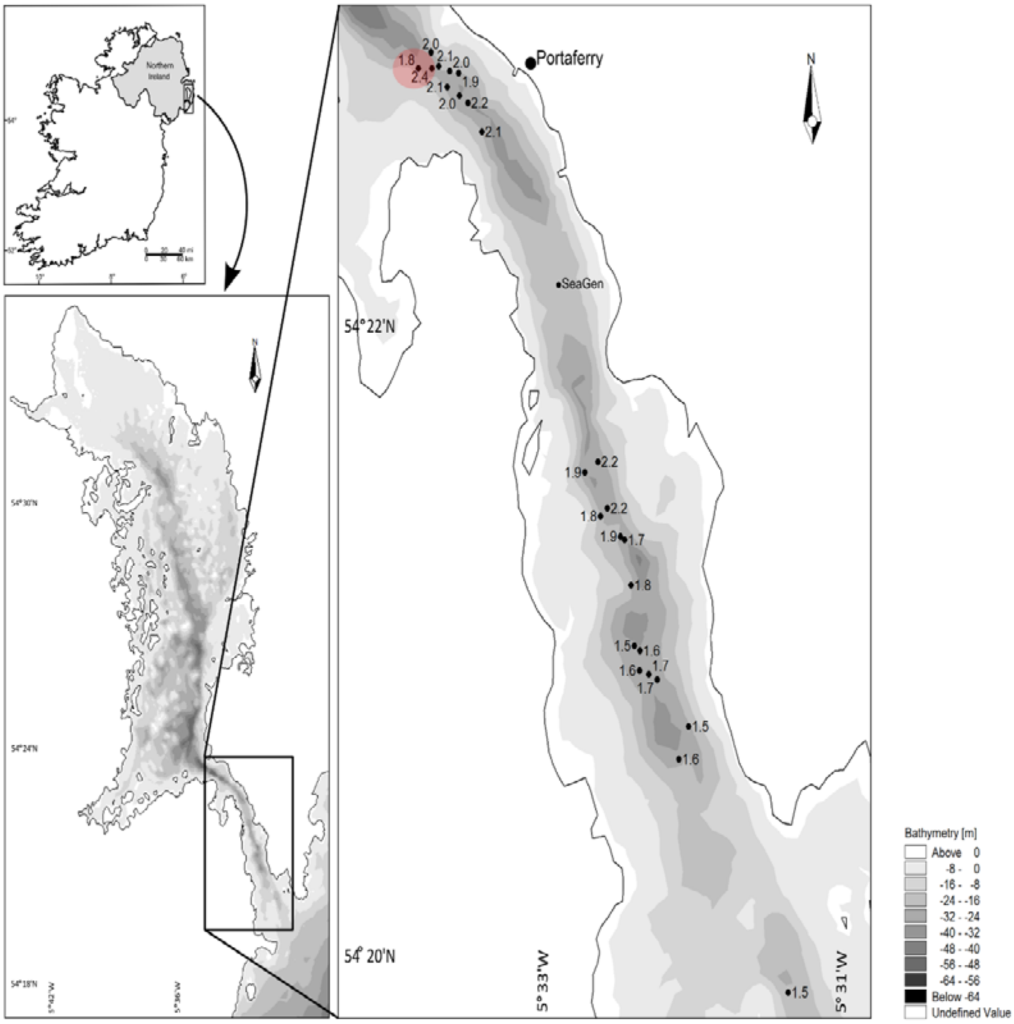

Figure 6: Map of the Narrows and the 25 sites identified for average current velocities.

Species	Av.Abund	Av.Sim	Sim/SD	Contrib%	Cum.%
Halichondria (Halichondria) panicea	2.3	9.13	1.21	23.18	23.18
Spirobranchus	0.96	6.13	1.55	15.56	38.74
Bryozoa	0.99	4.44	1.3	11.28	50.02
Alcyonium digitatum	1.57	4.08	0.61	10.36	60.38
Porifera	1.3	3.02	0.59	7.66	68.04
Hydrozoa	1.22	3.01	0.64	7.65	75.69
Obelia	1.36	2.83	0.47	7.19	82.88
Balanus	0.81	2.04	0.6	5.18	88.06
Alcyonidium diaphanum	0.98	1.56	0.37	3.96	92.02
Average Similarity: 34.4%					

Table 9: SIMPER table of results outlining the characterising fauna for the area surveyed.

A nested mixed model permutational analysis of variance (PERMANOVA) was used to test for an effect of velocity classes on epifaunal community structure. The random factor Site was nested in the fixed factor Vel (Velocity class). There was no significant effect of mean current velocity on epifaunal communities (See Table 10). The range of tidal velocities most relevant to tidal energy extraction technologies, in ecological terms, appear to be homogenous in their effects on epifaunal community structure. This accounts for the lack of response in epifaunal community structure to the velocities assessed in this study.

Source	df	SS	MS	Pseudo-F	P(perm)	Unique perms	P(MC)
Vel	8	1.08E+05	13457	0.89696	0.6637	9881	0.6678
Site(Vel)	16	2.40E+05	15003	13.343	0.0001	9791	0.0001
Res	500	5.62E+05	1124.4				
Total	524	9.10E+05					

Table 10: PERMANOVA table of results. There was a significant effect of Site(Vel), p=<0.0001.

Epifaunal communities within the Strangford Lough Narrows were found to be resilient to a significant range of high current velocities. This indicates that the communities present in tidal energy extraction sites could be capable of withstanding modified flow conditions within the turbulent wake of a turbine. In order to definitively assess this hypothesis, a modelled simulation of the turbulent wake created by a turbine would be required. Tests for correlations between high resolution spatial variation in epifaunal communities adjacent to the turbine and the modelled wake would then be necessary.

2.5 Case study 5

Author declaration: My contribution to this manuscript involved: Field work, all video still taxonomic analyses, all statistical analyses, all geospatial analyses and the writing of the document.

This Study was submitted to Marine Environmental Research on the 11th of November 2016:

FLOWBEC: Assessing spatial variation in an epifaunal community in relation to the turbulent wake created by a tidal energy turbine.

J.P.J O'Carroll, R.M. Kennedy, A. Creech, G. Savidge

Abstract

The effects of modified flow on epibenthic boulder reef communities adjacent to the SeaGen, the world's first grid-compliant tidal stream turbine, were assessed. The modified wake of the SeaGen was modelled, and the outputs were used in conjunction with positional and substrate descriptor variables, to relate variation in epibenthic community structure to the physical environment. An Artificial Neural Network (ANN) and Generalised Linear Model (GLM) were used to make predictions on the distribution of Ecological Status (ES) of epibenthic communities in relation to the turbulent wake of the SeaGen. ES was assigned using the High Energy Hard Substrate (HEHS) Index. Epifaunal community structure was highly variable and could not be attributed to environmental variables using an ANN or GLM. Spatial pattern in epifaunal community structure was detected when the study area was partitioned into three treatment areas: area D1; within one rotor diameter (16 m) of the centre of SeaGen, area D2; between one and three rotor diameters, and area D3; outside of three rotor diameters. Area D1 was found to be significantly more variable in terms of epifaunal community structure, bare rock distributions and EQR values. However, this influence does not extend beyond the device footprint.

For a detailed description of this study refer to the manuscript in Article Chapter V.

2.5.1 Case study 5 synopsis

This study is the first attempt at relating variation in epifaunal reef communities to the turbulent wake created by a tidal energy turbine. The study area exists in the Strangford Lough Narrows which is part of the Strangford Lough Natura 2000 site (See Fig. 7). The aim of this study was to produce a predictive model for the spatial extent of any potential benthic impacts associated with tidal energy extraction. This was done with the view to establishing a 'likely' disturbance radius for use in informing the strategic management of future, similar instalments. A high resolution drop-down video survey was carried out around the device and was used to derive percentage faunal coverage, substrate type distributions and the distribution of bare rock.

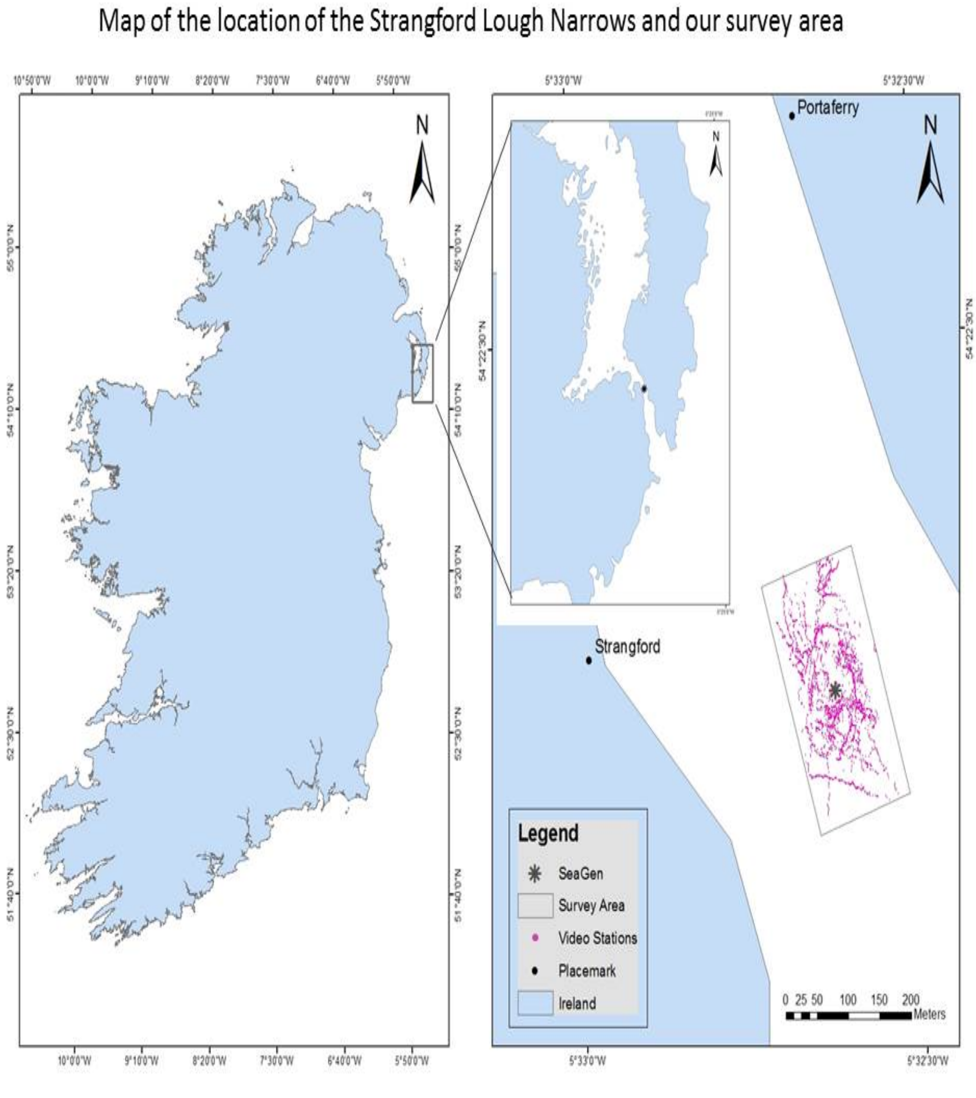

Figure 7: This figure shows the location of the Strangford Lough Narrows, our survey area, the position of the SeaGen and the georeferenced drop-down video stations.

A high resolution simulation of the turbulent wake created by the turbine was also modelled (Creech, 2014, Creech, in prep.). The HEHS index was applied to the data to test for a relationship between the distribution of ES and the modified ambient flow conditions. The results of this study will give good information on how tidal energy extraction affects epifaunal communities and on the effectiveness of the HEHS index and its potential for standardising tidal energy monitoring outputs globally.

The field survey was carried out over two legs, from 14th to 20th of May 2013 and from 1st to the 2nd of June 2013. The first leg involved a spatially high resolution drop-down video survey of the seafloor surrounding the SeaGen device. The second leg consisted of a scuba survey of the seafloor immediately beneath the device using the same drop down camera frame. The survey was conducted within a 300 m X 150 m rectilinear polygon (See Fig. 7) with the longer sides of the polygon running parallel to both E and W banks of the Narrows and the SeaGen device being positioned at the centre.

A high frame rate (50 frames per second) high definition camera system was used to obtain video footage of the seafloor around the device. A USBL (Ultra Short Baseline) system was mounted to the camera system and a differential geographic positioning system (DGPS) enabled its position to be georeferenced to sub one meter location accuracy.

Video transects were run during slack water on neap tides. A total of 20 drop-down video transects were surveyed. Three diver video surveys were carried out to ensure good complementary coverage of the seafloor in the immediate vicinity of the device.

Video analysis was carried out on stills and footage using VLC media player. The computer screen was divided into one hundred cells using a grid printed on acetate which was placed on the front of the monitor. The area of the seafloor captured by the camera system at any point was on average $50cm^2$, determined by scale lasers mounted on the camera frame. A grid with 100 cells was printed on an acetate sheet. Each cell covers approximately $0.5cm^2$ of substrate. This allowed for the estimation of percentage faunal cover and the estimation of substrate type distribution (e.g. percentage of gravel, cobbles, boulders and bare rock). Over 3,500 video stills were analysed for this study.

A computational fluid dynamics (CFD) simulation of a tidal turbine similar to the SeaGen device was developed, with a solid structure, and rotors modelled with the actuator line technique (Creech, 2014, Creech, 2016). The CFD model outputs used in the ecological analyses were time-averaged data for average current velocity (Vel) and turbulence intensity (Turb) at 1 m above the sea floor. The coordinates of each georeferenced video still station were inserted into the CFD model and used as specific simulation nodes so that hydrodynamic parameters could be accurately simulated for each point.

Artificial Neural Networks (ANNs) were constructed using the nnet package (Venables and Ripley, 2002) in the R statistical software environment v3.3.1 (R Development Core Team, 2014). Two ANNs were trained to make predictions on a binary response along the 'Moderate/Good' boundary and on five nominal responses; Bad, Poor, Moderate, Good, High. Six environmental predictor variables were used. Distance (m) from the turbine (Dist) was used as a continuous predictor variable. Hydrodynamic variables (Vel and Turb) were used as continuous variables. Boulder tops and bedrock were included as binary substrate type distribution predictor variables. A binary variable (Stream) was used to describe whether a station was upstream or downstream of the SeaGen. ANNs were unable to relate ES classifications produced by the HEHS index to environmental variables.

Epifaunal community structure was found to be highly variable in the survey area. SIMPROF analysis (Clarke et al., 2008) of square root transformed epifaunal abundance data created 120 significantly different groups based on Bray-Curtis similarities (Bray and Curtis, 1957).

BVSTEP analysis (Clarke and Ainsworth, 1993) was used to test if the distribution of a single species or a subset of species could explain pattern in multivariate community structure. This procedure was used to test for high rank correlations between the square root transformed faunal abundance data matrix and its own BC resemblance matrix. Weighted Spearman was used as the correlation coefficient.

A one way permutational analysis of variance (PERMANOVA; Anderson, 2008) was carried out on the zero adjusted transformed resemblance matrix using distance from the turbine as a fixed factor with three levels: D1; within 16 m (one rotor diameter) of the device, D2; between 17 m and 48 m and D3; more than 48 m away from the device. This allowed potential effects on multivariate epifaunal abundance data, bare rock distributions and EQR values to be assessed in a radial manner around the turbine.

PERMANOVA found the factor distance (D) to significantly affect all of the response variables (See Table 11). It was found that area D1 contained the lowest average EQR value than any other treatment level which fell below the critical 'Moderate/ Good' boundary. D1 also contained significantly more bare rock than the other treatment areas (See Table 12). PERMDISP analyses showed that multivariate epifaunal community structure was more variable in D1 than any other treatment area. Each treatment area was also significantly different in terms of its community structure, indicating that in D2 there may be intermediate state communities existing between the impacted D1 area and the greater survey area (See Table 13).

The impact detected by the HEHS index in area D1 exists between and immediately adjacent to the bottom quadrat legs of the SeaGen turbine. It was not possible to include this structure in the CFD model. It is possible that there is a scouring effect causing higher variability in faunal assemblages and low ES classifications in this area. This impact is not considered to pose a significant risk to the conservation status of the Strangford Lough Narrows.

(A) Bare rock

Source	df	SS	MS	Pseudo-F	P(perm)	Unique perms
D	2	4.571	2.2855	26.321	0.001	999
Res	1452	126.08	8.68E-02			
Total	1454	130.65				

(B) EQR

Source	df	SS	MS	Pseudo-F	P(perm)	Unique perms
D	2	4.6632	2.3316	197.63	0.001	997
Res	1452	17.13	1.18E-02			
Total	1454	21.794				

(C) Multivariate fauna

Source	df	SS	MS	Pseudo-F	P(perm)	Unique perms
D	2	20078	10039	7.9541	0.001	998
Res	1452	1.77E+06	1262.1			
Total	1454	1.79E+06				

Table 11: This table shows the results of PERMANOVA on (A) Bare rock distributions (B) EQR values and (C) Multivariate faunal community structure (Multivariate fauna). The fixed factor D had a significant effect on the response variables in all cases.

(A) BR

D1			
Average sqrd distance = 0.14			
Species	Av.Value	Av.Sq.Dist	Sq.Dist/SD
%BR	0.476	0.141	0.54
D2			
Average sqrd distance = 0.09			
Species	Av.Value	Av.Sq.Dist	Sq.Dist/SD
%BR	0.285	9.44E-02	0.47
D3			
Average sqrd distance = 0.08			
Species	Av.Value	Av.Sq.Dist	Sq.Dist/SD
%BR	0.294	8.36E-02	0.47

(B) EQR

D1			
Average sqrd distance = 0.08			
Species	Av.Value	Av.Sq.Dist	Sq.Dist/SD
EQR	0.567	8.39E-02	0.5
D2			
Average sqrd distance = 0.05			
Species	Av.Value	Av.Sq.Dist	Sq.Dist/SD
EQR	0.694	4.73E-02	0.37
D3			
Average sqrd distance = 0.04			
Species	Av.Value	Av.Sq.Dist	Sq.Dist/SD
EQR	0.713	3.75E-02	0.33

Table 12: This table summarises six SIMPER tables of results. SIMPER analysis was carried out to outline the compositional differences between treatment levels of the factor D based on; (A) Bare rock (BR) distributions and (B) EQR values. Percentage contribution and cumulative contribution were omitted from the tables as both equalled 100% in each case.

Multivariate fauna

Groups	t	P(perm)	Unique perms
D2, D1	3.2499	0.001	999
D2, D3	1.9222	0.012	999
D1, D3	3.5571	0.001	999

Table 13: This table shows the results of PERMANOVA Pair-wise comparisons between the three treatment levels of the fixed factor D on Multivariate faunal community structure. This table shows that all the treatment levels were significantly different in terms of their epifaunal community structure. t= The test statistic, P(perm)= P values obtained through permutations, Unique perms= amount of permutations, P(MC)= P value obtained by Monte-Carlo test.

2.6 Case study 6

This study is in preparation and will be submitted to Estuarine and Coastal Shelf Science January 2017:

Author declaration: My contribution to this manuscript involved: Carrying out all taxonomic analyses for in-house surveys, majority of statistical analyses, all GIS, and the writing of the manuscript.

Using Artificial Neural Networks to predict the spatial distribution of EUNIS biotopes in a large embayment

Jack P.J. O'Carroll, Robert Kennedy, Lei Ren, Stephen Nash, Michael Hartnett, Colin Brown

Abstract

In this study, feed-forward artificial neural networks (ANNs) were used to make predictions on the spatial distributions of EUNIS level 4 subtidal sediment biotopes in the largest marine embayment in Ireland, Galway Bay. A cost effective modelling approach was developed with the aim of making it applicable in offshore, MSFD benthic mapping exercises. The INFOMAR (Integrated Mapping of Irelands Marine Resource) initiative has acoustically mapped and classified a significant proportion of Irelands Exclusive Economic Zone (EEZ). As much publically available data as possible was incorporated in our model which included depth and INFOMAR acoustic seabed data, transitional water boundaries and grab survey data. Targeted grab surveys were carried out by the authors to fill any spatial gaps in the data. A hydrodynamic model was developed in-house and is a high resolution 3D numerical model of Galway Bay. The Ecological Status (ES) of the subtidal sediments of Galway Bay was also assessed.

The ES of Galway Bay was mostly 'High' with some stations being classed as 'Good'. To make predictions on the distribution of EUNIS biotopes optimal models were determined using multinomial modelling techniques prior to making predictions using ANNs in R. Optimal models used a combination of salinity, proximity to reef, depth and a sediment descriptor as predictor variables.

ANNs that used observed sediment classes as predictor variables could predict the distribution of biotopes 67% of the time, compared to 63% for ANNs using acoustic sediment classes. Acoustic sediment ANN predictions were affected by local sediment heterogeneity, and the lack of a mixed sediment class. Within

Galway Bay, INFOMAR data can be used as a surrogate for traditional sediment predictor variables and only result in a small reduction in total correct model predictions.

For a detailed description of this study refer to the manuscript in Article Chapter VI.

2.6.1 Case study 6 synopsis

This is the first study to use INFOMAR sediment acoustic classifications as predictor variables in a predictive spatial model for EUNIS biotope distributions. INFOMAR (Integrated Mapping for the Sustainable Management of the Irelands Marine Resource) is a mapping initiative that has acoustically mapped and classified a significant proportion of the seafloor (~125,00 km^2) of Irelands EEZ. One of the aims of this study was to develop an approach for predicting the spatial distributions of EUNIS level 4 biotopes in Galway Bay in a manner that is cost-effective and applicable in the offshore environment. This would be a useful conservation tool considering the legislative requirements of the MSFD. We also assess the ES of Galway Bay using the IQI, and demonstrate the IQI's potential for use in assessing the status of benthic ecosystem integrity under the MSFD.

Grab data for this study was gathered from multiple sources. Thirteen stations of publically available grab data were used in analyses. The remaining seventy seven grab samples were collected and analysed by the authors. Bathymetric data for Galway Bay was downloaded for free from the INFOMAR website http://maps.marine.ie/infomar/. Transitional water boundaries were downloaded for free from http://gis.epa.ie/Envision/.

Cost effective techniques were adopted such as sediment subsampling and the classification of sediments into textural groups by experts in the field. This was done to demonstrate the potential these methods have to maximise survey efforts within limited monetary budgets.

Eleven of the publically available grab stations were sampled using two replicate Van Veen grabs for fauna, and a separate Van Veen grab for sediment PSA. The remainder of the grab data was collected by taking one Day grab sample per station, and each grab was subsampled for sediments. In all cases, faunal samples were sieved on a 1 mm mesh sieve, fixed with 10% buffered formalin and preserved in 70% alcohol. The taxa were then identified to species level where possible.

Associated sediment data for each faunal sample was derived from textural groups produced by GRADISTAT (Blott and Pye, 2001) in the case of public data sources and from field descriptions in the case of in-house surveys. Textural group sediment descriptions were then transformed into EUNIS sediment classifications *sensu* Long et al., (2006) and used as ordinal predictor variables in statistical analyses. EUNIS sediment classifications will hereinafter be referred to as 'observed sediment' classifications

Six level 4 EUNIS biotopes were found to exist within Galway Bay by using the techniques outlined by Conor et al., (2004): 'Circalittoral Coarse Sediment' (SS.SCS.CCS), 'Circalittoral mixed sediment' (SS.SMx.CMx), 'Subtidal mixed sediments in varying salinity' (SS.SMx.SMxVS), 'Infralittoral Sandy Mud' (SS.SMu.ISaMu), 'Circalittoral fine sand' (SS.SSa.CFiSa) (See Figure 8). The CCS biotope was omitted from predictive models as it contained only one station.

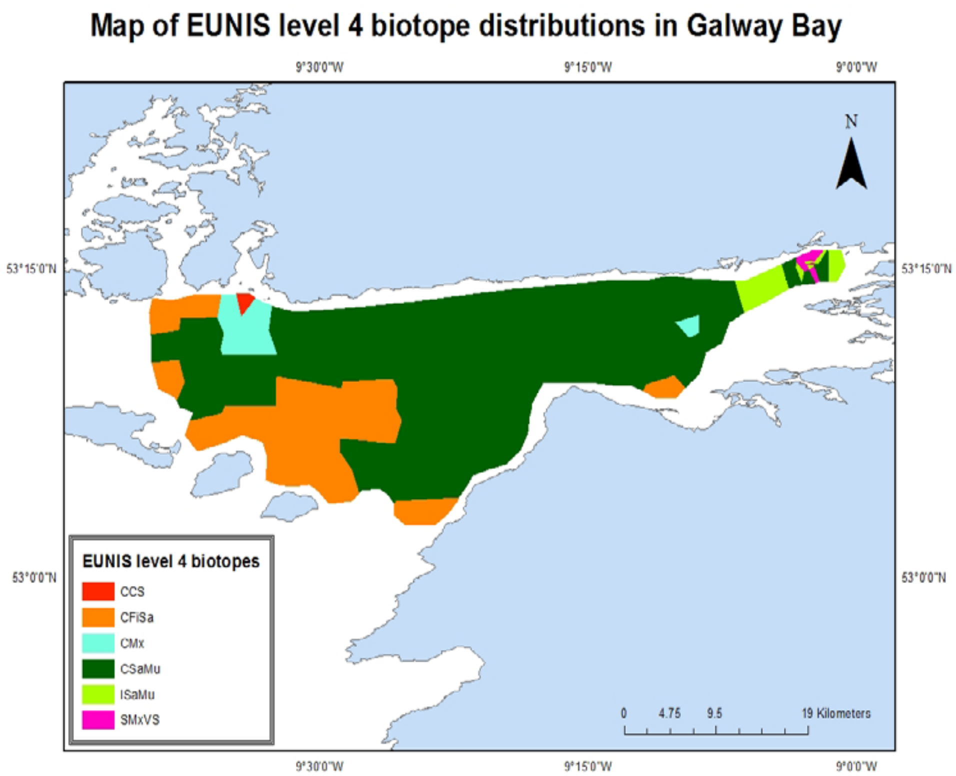

Figure 8: Map of the distributions of EUNIS level 4 biotopes in Galway bay.

2. Case studies

The three INFOMAR acoustic sediment classes fit into grain size intervals; muds to fine sands (9-4 phi intervals or 2-63 µm), fine sands to medium sands (4-1 phi intervals or 64-449 µm) and coarse sands to gravel (1- (-1) phi intervals or 0.5-2 mm). INFOMAR sediment classifications will hereinafter be referred to as 'acoustic sediment' classifications. Sediment textural groups were reclassified to correspond to the acoustic sediment classes in two ways. In the first, all textural groups with >5% gravel were classified as coarse sand to gravel (See Figure 9A). In the second, all textural groups with >30% gravel were classified as coarse sand to gravel (See Figure 9B). Kappa analyses showed the agreement between the scaled textural groups and acoustic sediment classes to be poor, with the 30% gravel boundary classification having the highest agreement (kappa value= 0.1, p= 0.291). This poor agreement is due to the different methods used to derive observed and acoustic sediment classifications. Sediment textural group classifications are derived through ternary classification which is based on ratios of three sediment components (Sand, Mud and Gravel). Also grabbing sediments from the seafloor may record sediment heterogeneity that acoustic swathe methods can miss.

(A) (B)

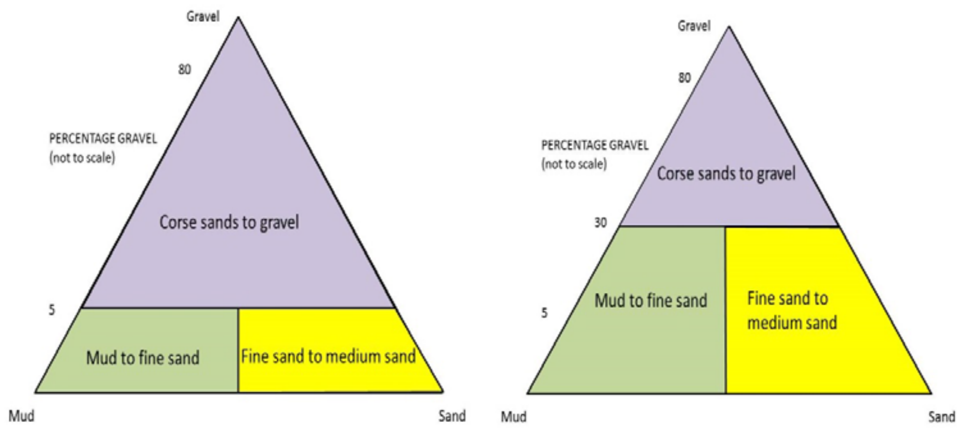

Figure 9: Two ternary classification triangles that have been modified to demonstrate how textural groups were classified as INFOMAR classes. (A) Triangle showing how textural groups were classified as INFOMAR classes along the 5% gravel boundary. (B) Triangle showing how textural group classifications were classified as INFOMAR classes along the 30% gravel boundary.

The majority of stations were assigned an ES of 'High' with only one station being classified as 'Good'. The putative impact sources in Galway Bay such as the dredge channel, treated sewage outfall pipe and dredge disposal area all exist at depths shallower than 10 m (O'Reilly et al., 2006). All stations in our data exist at depths greater than 10 m and occur at distances from these putative impact sources that are large enough to avoid significant deleterious impacts.

Multinomial models were trained with the entire dataset and were used to select the optimal models prior to making predictions using ANNs. Multinomial models were constructed using the nnet package (Venables and Ripley, 2002) in the R statistical software environment v3.3.1 (R Development Core Team, 2014). Predictor variables used in optimal models were a combination of salinity, depth, proximity to reef and a sediment descriptor. Multinomial models could correct classifications 80% of the time when observed sediment classes were used, compared to 73% when acoustic classes were used. Hydrodynamic variables never significantly contributed to model fit (measured as total correct predictions).

Two ANNs were trained with the same training dataset. ANNs were constructed using the nnet package (Venables and Ripley, 2002) in the R statistical software environment v3.3.1 (R Development Core Team, 2014). Both had two hidden layers, one used observed sediment classes (observed sediment ANN), and the other used acoustic sediment classes as sediment predictor variables (acoustic sediment ANN). The observed sediment ANN could make predictions 67% of the time, compared to 63% of the time for the acoustic sediment ANN (See Table 14). The observed sediment ANN also showed the best agreement with observed biotope distributions (Kappa value= 0.34, p <0.001). Acoustic sediment ANN predictions showed slightly poorer agreement with the observed distributions of biotopes (Kappa value= 0.24, p <0.0252). Acoustic ANN predictions were affected by local sediment heterogeneity, and the lack of a mixed sediment class. Both ANNs were affected by small size of training the training dataset.

(A)	Fitted				
Given	**1**	**2**	**3**	**4**	**5**
1	26	1	0	0	3
2	2	2	0	0	0
3	2	0	0	0	0
4	1	2	0	0	0
5	5	0	1	0	1

(B)	Fitted				
Given	**1**	**2**	**3**	**4**	**5**
1	22	6	0	0	2
2	1	2	0	0	1
3	1	1	0	0	0
4	0	0	0	3	0
5	3	0	0	0	4

(C)	No. of stations per biotope	No. of correctly predicted stns. per biotope	% correct predictions per biotope	Total correct predictions	Total % correct predictions
SS.SMu.CSaMu	30	26	86.6	29	63
SS.SMu.ISaMu	4	2	50		
SS.SMx.CMx	2	0	0		
SS.SMx.SMxVS	3	0	0		
SS.SSa.CFiSa	7	1	14.3		
Total	46				

(D)	No. of stations per biotope	No. of correctly predicted stns. per biotope	% correct predictions per biotope	Total correct predictions	Total % correct predictions
SS.SMu.CSaMu	30	22	73. 3	31	67.4
SS.SMu.ISaMu	4	2	50		
SS.SMx.CMx	2	0	0		
SS.SMx.SMxVS	3	3	100		
SS.SSa.CFiSa	7	4	57.1		
Total	46				

Table 14: Results of ANN predictions. (A) Goodness of fit table for the INFOMAR ANN. (B) Goodness of fit for the EUNIS ANN. (C) Performance table for the INFOMAR ANN. (D) Performance table for the EUNIS ANN. **1**= SS.SMu.CsaMu, **2**= SS.SMu.ISaMu, **3=** SS.SMx.CMx, **4**= SS.SMx.SMxVS, **5**= SS.SSa.CFiSa.

Within Galway Bay, acoustic sediment classes appear to have the potential to be used as surrogates for observed sediment classes in predictive models. The marginally poorer performance of acoustic sediment classes as predictor variables is arguably acceptable when the small difference in prediction rates and the monetary savings that could be made. Further research is required to compare the effectiveness of observed versus acoustic sediment classification based predictive models in the offshore environment.

3. Discussion

3.1 The flexibility of EQRs and their effectiveness as monitoring tools in new environments

The growing use of the coastal environment by humans is leading to an intensification of pre-existing anthropogenic pressures, and the proliferation of new ones (Halpern et al., 2008, Vitousek et al., 1997). Anthropogenic activities such as aquaculture and marine renewable energy production have the potential to provide food and energy for growing human populations in a sustainable manner. The potential effects of these activities must be definitively assessed prior to expanding these industries in the Irish coastal environment.

Case studies 1, 2, 3, 5 and 6 (Forde et al., 2015, O’Carroll et al., 2016, O’Carroll et al., 2017a, O’Carroll et al., 2017b in review, O’Carroll et al., 2017c in review) demonstrate the potential WFD benthic EQRs have to standardise benthic monitoring across fundamentally different habitat types in relation to varying modes of anthropogenic pressures.

Case studies 1 and 2 showed the IQI to be an effective tool for monitoring the associated impacts of intertidal bivalve trestle cultivation (Forde et al., 2015, O'Carroll et al., 2016). This adds to the growing literature that demonstrates the flexibility and potential the IQI has as a monitoring tool in new environments outside of sublittoral fine sediments (Fitch et al., 2014, Fitch and Crowe, 2010).

The successional model outlined by Pearson and Rosenberg (1978) has been central to the development of EQRs such as IQI and Multivariate AMBI (Azti Marine Biotic Index) that incorporate species tolerances to stress (Grall, 1997, Borja et al., 2000). The AZTI Marine Biotic Index (AMBI) AMBI is a single metric index which is calculated based on the proportions of five Ecological Groups (EG) to which benthic species are allocated depending on their tolerance to disturbance (Borja et al., 2007; Muxika et al., 2007). The assignment of species to EG in the AMBI is based on the extensive literature describing North Atlantic species and, where the existing literature is lacking, consensus expert judgement (Teixeira et al., 2010). Based on AMBI index values benthic communities are classified as undisturbed, slightly disturbed, moderately disturbed, heavily disturbed or extremely disturbed (Muxika et al., 2007). The IQI includes AMBI as a metric and this addresses the requirement to account for 'the proportion of sensitive taxa' in EQRs developed in response to the WFD. In case studies 1 and 2, applying the IQI allowed for the detection of deleterious change in relation to trestle cultivation in a readily interpretable and comparable way across multiple sites.

Johnson (1971, 1972) described the sedimentary habitat as a temporal and spatial mosaic "parts of which are at different levels of succession… in this view the community is a collection of relics of former disasters". The Pearson and Rosenberg (1978) model is a development on this paradigm, and is described in relation to oxygen depletion. Sousa (1979) described the surface of a boulder as "a patch of habitat which differs in size and age from that of neighbouring boulders". This is analogous to the description of the successional patchiness of the sedimentary habitat made by Johnson (1971, 1972). In Case study 3, the successional model depicted in Figure 4 is a development on Sousa's description of boulder top communities described in relation to physical disturbance. The model proposed in Figure 4 was incorporated into a benthic EQR that meets all the criteria outlined by the WFD. The 'level of diversity and abundance of invertebrate taxa' is addressed by the Shannon Weiner index (H') (Shannon and Weaver, 1949) and number of species (S). The 'proportion of disturbance sensitive taxa' is addressed by the Massive/Crust ratio.

Central to the proposed successional model in Figure 4 is the assumption that on tide swept, stable reef substrates, massive taxa are 'disturbance sensitive'. Both the massive and encrusting taxa within the Narrows are subjected to high amounts of scouring by suspended sediment twice a day. The spatial extent of the hydroid and sponge dominated community within the Narrows shows that both massive and encrusting taxa are inherently tolerant to these high levels of physical stress. However, the massive taxa exhibit morphological characteristics that are not as robust to extreme physical disturbance events as encrusting taxa.

Massive taxa protrude from the substrate into the water column. They also have a small proportion of their body mass in contact with the substrate. Having such a small point of attachment could result in disturbance events such as brief periods of very high frictional flow (Vogel, 1996), overturning, physical abrasion via mobile gravel cobbles or scouring by suspended sediment (Palmer and Palmer, 1977, Daly and Mathieson, 1977) being catastrophic for the entire animal/colony if the holdfast is damaged.

In the case of encrusting taxa, their morphology reduces the amount of surface area that can be subjected to frictional stress by flow (Vogel, 1996). If an encrusting taxa covers an area greater than that which is affected by a disturbance event, small patches of the animal will remain to recolonize the substrate (Osman, 1977). This pertains to the traditional successional paradigm that a disturbed area will host higher abundances of stress tolerant opportunistic taxa then sensitive taxa (Pearson and Rosenberg 1978, Sousa 1979a, 1979b, Morris and Wood, 1989, Glémarec and Hily, 1981, Gral and Glémarec, 1997).

A reef's 'surface heterogeneity' refers to its physical and biological topographic features. The surface of a patchy reef substrate is less heterogeneous than a fully colonised undisturbed reef. This is because encrusting animals and patches of bare rock have less surface area than massive taxa. The HEHS index allows a reef substrate and its resident epifaunal community to be viewed as a single entity that will reduce its surface heterogeneity in response to frictional disturbance conditions. The point at which the homogenisation of a boulder's surface heterogeneity can be considered a significant impact can only be decided by expert judgement and observations from within the habitat in question (Muxika et al., 2007).

Case studies 3 and 5 demonstrate that the HEHS index responds in an intuitive manner to changes in established measures of sensitivity for high energy, animal dominated, subtidal reefs. Case Studies 1 and 2 showed the IQI to be effective at detecting deleterious change in highly variable intertidal mobile sand communities. These are good examples of the ability of benthic EQRs to detect deleterious change within highly variable habitats that univariate and multivariate analyses can miss. Traditional multivariate and univariate methods can produce results that can be difficult to interpret in terms of disturbance effects (Underwood, 1991, Underwood, 1994). Our results show that using benthic EQRs to compare communities and habitats based on the relative proportions of disturbance sensitive taxa is a more appropriate method of anthropogenic monitoring than the use of species abundances and univariate diversity indices.

3. Discussion

Monitoring exercises aimed at assessing the conservation status of SACs are required to establish whether or not the conservation status for the area is favourable or unfavourable. As demonstrated in Case Studies 1, 2, 3 and 5, the IQI and HEHS index can be adjusted to give a binary output at the critical Good/Moderate boundary which can be used to determine the habitat quality element of conservation status.

The MSFD also requires that any anthropogenic activities likely to affect the integrity of the seafloor are to be monitored. Case study 6 is an example of how an EQR such as IQI could be applied in the offshore environment in an assessment of the areas environmental status. As in Case Studies 1, 2, 3 and 5, the binary output along the Moderate/Good boundary could be used to determine if an area has a GES or not. The survey methods used in Case study 6, such as sediment subsampling can significantly affect the performance of EQRs (Kennedy et al., 2011). However, once taken into account, these issues can be circumvented by adjusting the class boundary cut-off points of the EQR appropriately.

Currently, EQRs have been shown to be amenable to modification (Forde et al., 2015, Forde et al., 2013), transferable across large geographical distances (Forde et al., 2013), capable of detecting a range of anthropogenic impacts (Rhoads et al., 1978, Borja et al., 2000, Van Dalfsen et al., 2000, Borja et al., 2003, Muxika et al., 2005, Bouchet and Sauriau, 2008, Borja et al., 2009b, Kennedy et al., 2011, Forde et al., 2015) in various environments (Van Hoey et al., 2007, Fitch and Crowe, 2010, Kennedy et al., 2011, Forde et al., 2013, Fitch et al., 2014, Forde et al., 2015, Barry, 2013).

Applying benthic EQRs to monitoring exercises in Natura 2000 sites and MSFD waterbodies could strengthen their conservation objectives. There is potential for EQRs to facilitate the application of standardised benthic monitoring protocols across the HD, BD, WFD and MSFD. Adopting a standard monitoring approach would allow for the integration of all the Directives in terms of benthic habitat monitoring as required under the MSFD. However, before applying benthic EQRs to designated sites outside the remit of the WFD it is essential to determine the suitability of multimetric tools for assessing the putatively disturbed communities or habitats under investigation. The unvalidated application of benthic EQRs in areas for which they were not originally developed has been criticised (Forde et al., 2013, Quintino et al., 2006). Principally because in such instances there is a risk of flawed management decisions being made. EQRs should be chosen based on the type of disturbance to be assessed and with a knowledge of the ecological basis and limitations of the tool in question and, in particular, the adequacy of the data used to derive the tool (Quintino et al., 2006, Kroncke and Reiss, 2010, Kennedy et al., 2011, Forde et al., 2013).

Different EQRs can potentially produce different outputs when applied to the same datasets. An example of this is the study carried out by Fitch and Crowe (2010). They used a range of EQRs to assess the effects of organic enrichment on intertidal habitats. The EQRs showed good agreement in terms of relative ranking of EQR values, but the agreement between ES classifications was poor. Intercallibration studies are required to circumvent these issues (Muxika et al., 2007). Intercallibration exercises typically involve adjusting EQR class boundaries so as to increase agreement between EQRs in terms of ES classifications (Forde et al., 2013, Bouchet and Sauriau, 2008) with particular importance attached to agreement between indices measures around the ‘‘Good/Moderate’’ boundary (i.e. the critical WFD boundary) (e.g. Borja et al., 2007). The implications for management bodies making decisions based on EQRs that have not been intercallibrated are likely to be financially costly. If an EQR classifies a waterbody as Moderate or less, when it otherwise would not post intercallibration, unnecessary costs for remedial management actions would be incurred when improving the ES from Moderate or less to Good or higher (Quintino et al., 2006).

3.2 The effectiveness of hydrodynamic predictor variables at small and medium spatial scales

The influence of ocean currents on the physical and biological elements of the marine ecosystem has led to the utilisation of hydrodynamic models in anthropogenic monitoring (Grant et al., 1995) and habitat mapping (Dutertre et al., 2013, Christensen et al., 2009) exercises. Given the influence of hydrodynamic regimes on the distributions of biological communities and single species populations, understanding the relationships between them is critically important from a conservation biology perspective (Treml et al., 2008).

3. Discussion

Animal-flow interactions have been shown to be highly variable (Okamura, 1984, Okamura, 1985, Okamura, 1988, Vogel, 1996, Vogel, 1994). Further investigation into this relationship is required due to the expansion of industries such as tidal energy extraction. The benthic footprint (Miller et al., 2013) of a tidal energy turbine is the physical area that is directly or indirectly effected by the process of tidal energy extraction. Effects such as increased turbulence, alteration of sedimentation regimes and the scouring of benthic communities are suggested to be the most prominent threats to the benthic ecosystem associated with tidal energy extraction (Miller et al., 2013, Shields et al., 2009, Shields et al., 2011, Gill, 2005). Currently, the most immediate threats to benthic assemblages within tidal energy extraction sites are those posed by single operational devices. These impacts are likely to be localised (within 30 m of the turbine) and a result of increased physical disturbance. When a tidal stream turbine is operational a turbulent wake is created downstream of the device. This turbulent wake has the potential to negatively impact benthic communities up to the point where the wake mixes back into the bulk flow, restoring homogenous conditions (Polagye, 2011). Within a turbulent wake, modified flow conditions may result in the scouring or removal of fauna due to accelerated flow around large bottom structures and substrates, or through the mobilisation of substrates (Daly and Mathieson, 1977, Palmer and Palmer, 1977, McGuinness, 1984, McGuinness, 1987b, McGuinness, 1987a, McGuinness and Underwood, 1986).

Case study 4 (Kregting, 2016) assessed the effects of a range of current velocities typically found in a tidal rapid environment on the structure of epibenthic communities. Epibenthic community structure in Strangford Lough was found to be unaffected by significant variation in average current velocities. This result indicates that the removal of energy from a tidal rapid environment by an operational tidal energy turbine, resulting in the reduction of mean velocity speeds from 2.4 ms^{-1} to 1.5 ms^{-1}, will not deleteriously affect epibenthic community structure.

The removal of energy from a tidal rapid environment due to tidal energy extraction will result in a change in flow conditions other than mean current velocity, such as turbulence (Miller et al., 2013, Polagye, 2011). Case study 5 found the simulated turbulent wake of an operational crossbeam, twin rotor tidal stream turbine to have no relationship with spatial variation in epifaunal community structure, up or downstream of the device.

The results of Case Studies 3, 4 and 5 found that epifaunal communities in Strangford Lough Narrows do not significantly change over relatively small spatial scales. When significant temporal change occurred in community structure, it occurred equally throughout the Narrows. The lack of significant spatial gradients in community structure indicates that the physical environment, in ecological terms, is homogeneous. Homogeneous in this case, is a relative term. When compared to low energy subtidal sediment habitats, reef dominated tidal rapids are highly variable and topographically heterogeneous.

The opportunistic faunal dominated community of the Narrows is well adapted for life in highly stressful conditions. The naturally occurring gradients of stress and levels of physical disturbance are not capable of permanently removing these communities, but introduce high amounts of localised variability in the distributions of the dominant epifauna. Other investigations have also found colonisation patterns on boulders to be highly variable and difficult to predict (Chapman, 2002a, Chapman, 2007). Within a dynamic physical environment such as the Narrows, physical processes are likely to override interspecific interactions in structuring the epifaunal community (Berlow, 1997). These physical processes are likely to be driven by abrasive forces that are a function of hydrodynamic conditions (Osman, 1977, Sousa, 1979a, Chapman, 2002a). When viewed in this manner, the epifaunal community of the Strangford Lough Narrows and its governing physical environment can be regarded as a homogenously variable entity.

Case study 6 used the outputs of a 3D hydrodynamic model of Galway Bay as predictor variables in ANNs. Depth averaged current velocity did not significantly contribute to model fit in any case. This apparent lack of correlation between current velocity and benthic communities is a result of the sedimentary habitat surveyed existing in a range of current velocities that, in ecological terms, are homogeneous. Reefs were not surveyed as part of this study. It is possible that if biotope response variables for reef communities were included, hydrodynamic variables may have contributed more to the models. Our models found factors such as depth, sediment type, salinity and proximity to reefs to influence infaunal community distributions in Galway Bay. This study produced similar results to those of case studies 4 and 5 in terms of the poor performance of hydrodynamic parameters in explaining spatial variability in benthic communities.

Hydrodynamic variables have been shown to significantly contribute to models that make predictions on benthic biotopes which cover a range of seafloor morphologies within a survey area (Christensen et al., 2009). The area surveyed by Christensen et al. (2009), in places exhibited current velocities of up to 2.5 ms^{-1}. Within Galway Bay maximum current velocities peaked at 1.6 ms^{-1}, with this peak occurring over reef. Dutertre et al., (2013) also found significant correlations between hydrodynamic variables and variation in the spatial distributions of benthic communities. Dutertre et al., (2013) sampled a survey area that extended over 150 km of coastline, and reported hydrodynamic conditions to be increasingly complex in close proximity to the islands that existed within the survey area. Significant estuarine plumes held significant influence over hydrodynamic conditions in parts of the survey area also.

The survey areas covered in Case Studies 4, 5 and 6 are relatively homogenous when compared to those surveyed in existing literature that found significant correlations between benthic community structure and hydrodynamic variables (Christensen et al., 2009, Dutertre et al., 2013). This prevented hydrodynamic parameters from contributing to the explanation of variance of the spatial distribution of benthic communities in our models. Our results show that the effectiveness of hydrodynamic variables in predictive modelling exercises is not only scale dependant, but also dependant on significant habitat heterogeneity existing in the area surveyed.

3.3 Cost effective approaches to benthic conservation biology

Benthic EQRs have the potential to facilitate time and cost effective monitoring programs within designated waterbodies. Forde et al, (2013) showed that the Multivariate AMBI (Azti Marine Biotic Index: Borja et al., 2000) EQR can be applied to data with taxonomic levels higher than species and still produce useful results. Other literature shows that reduced taxonomic efforts such as using only some of the ecological groups present in an area does not significantly affect the performance of benthic EQRs (Kennedy et al., 2011).

3. Discussion

The results of Case Study 5 indicate that there is potential to reduce taxonomic efforts to a subset of dominant fauna when monitoring in a tidal energy extraction site. The three dominant taxa *Tubularia indivisa, Sertularia argentea, Halichondria panicea* and the morphological type description 'mixed faunal turf' equally represent the two ecological functional group in the 'Massive: Crust' ratio used in the HEHS index. There was a reduction in the proportion of massive hydroid fauna relative to encrusting fauna and an increase in the percentage of bare rock in treatment area D1 (within 1 rotor diameter of the turbine). This change in community structure pertains to the predictions made in the successional model used in the HEHS index. Our results suggest that it may be possible to detect spatial effects of tidal energy extraction on epifaunal reef communities by assessing dominant faunal types only. This could significantly save on associated costs of monitoring efforts by reducing time spent analysing video stills.

One of the main issues facing EU Member States with legislative obligations to fulfil under the MSFD, is the significant cost associated with the extensive mapping of the benthic environment. Case Study 6 presents a predictive modelling approach that uses some publically available data sources and time-efficient survey techniques that can make good predictions on the distribution of EUNIS level 4 biotopes in Galway Bay, Ireland.

Multinomial models trained with the entire dataset could make correct reclassifications approximately 75% of the time using both acoustic and observed sediment predictors. Predictions made by ANNs that were trained with half of the data made correct predictions on biotopes between 63% (acoustic sediment ANN) and 67% (observed sediment ANN) of the time. These results show the effect of small sample size on model performance, and indicate that a higher number of sampling stations could have enabled ANNs to perform better at making predictions (Hernandez et al., 2006). Multinomial models were principally used to assess which combination of environmental predictor variables form optimal predictive models for subsequent use in ANNs.

Predictive models that used observed sediment classes as predictor variables performed consistently better than those using acoustic sediment classifications. The significant difference between the observed sediment and acoustic multinomial models only amounted to a 7% difference in total correct predictions.

3. Discussion

The observed sediment ANN also showed better agreement with observed biotope distributions and this agreement was highly significant. The difference in ANN correct predictions was only ~5%. These are small margins of difference when considering the monetary savings that can be made through the use of publically available sediment data with significant spatial coverage.

Our results show that INFOMAR sediment classifications could be used as a surrogate for observed sediment classes when predicting the distributions of biotopes in Galway Bay, and that doing so will only incur a ~6% reduction in total correct predictions. This demonstrates the potential in INFOMAR data for use in large scale benthic habitat mapping, which has major implications for legislative requirements of the MSFD. Other similar studies have also found acoustic signal derivatives to be effective environmental predictor variables for the spatial distribution of biotopes (Buhl-Mortensen et al., 2009, Ierodiaconou et al., 2011). This study adds to the growing literature which demonstrates the potential of acoustic sediment classifications for use in large spatial scale conservation efforts (Brown et al., 2011, Brown and Blondel, 2009, Buhl-Mortensen et al., 2009, Callaway et al., 2009, Cook et al., 2008, Ehrhold et al., 2006, Dolan et al., 2008, Dolan et al., 2009).

Within the innermost part of the Bay sediment type was variable and dominated by sands and mixed sediments. This variability is reflected in the spatial distributions of the ISaMu, MxVS and CSaMu biotopes in that small area. Acoustic classifications classified the innermost bay area as being largely mud to fine sands and fine sands to medium sands. This mismatch between observed and acoustic sediment classes could be a result of local heterogeneity in sediment distributions, which can cause acoustic swathe methods to miss fine resolution information on the seafloor (Brown et al., 2011, Brown and Collier, 2008). The lack of a mixed sediment class in the acoustic classifications is likely to have been a factor in its poor performance when making predictions on the mixed sediment biotopes.

The local scale heterogeneity of biotope distributions in the innermost bay area is also likely to have had an effect on the performance of acoustic sediment models. Sediment classifications made on sediment subsampled from faunal grabs will have a higher affinity for the biological community present (Somerfield and

Clarke, 1997) compared to an acoustic classification that is based on the mean sediment type within an area. The affinity between field sediment observations and the faunal samples enabled the observed sediment models to distribute their total correct predictions more evenly across the biotopes than the acoustic sediment models.

Heterogeneity in sediment distributions coupled with the intrinsically different procedures involved in deriving EUNIS or textural group sediment classes and INFOMAR acoustic sediment classifications had an effect on the agreement between the two. Textural groups are derived using the Folk and Ward (1957) ternary triangle. EUNIS sediment classes are derived through the use of a simplified Folk and Ward (1957) ternary classification triangle *sensu* Long (2006). This technique produces sediment classes that are based on grain size distribution ratios. The three INFOMAR acoustic sediment classes fit into grain size intervals; muds to fine sands (9-4 phi intervals or 2-63 µm), fine sands to medium sands (4-1 phi intervals or 64-449 µm) and coarse sands to gravel (1- (-1) phi intervals or 0.5-2 mm).

The composition of shallow sediments such as those in the innermost bay are temporally variable as a result of natural disturbance events such as sediment resuspension due to storm activity (Gray, 1981, O'Carroll et al., 2016). Acoustic classifications may have been made at a time when sediments in the area were more homogenous, leading to the mismatch between observed and acoustic sediment classifications.

The good level of correct predictions made by the observed sediment models suggests that subsampling sediment from grabs did not significantly affect sample affinities or the integrity of the faunal samples to any significant degree (Somerfield and Clarke, 1997). The motivation for single grab survey designs include cost reduction and the rationale that PSA distributions will have more affinity with macrofaunal distributions, allowing for their communities to be classified with increased confidence (Forde et al., 2012). This approach has been shown to significantly reduce the number of species in a sample. This has a negative effect on the ES assigned to a sample (Forde et al., 2012). Once this effect is taken into account prior to a monitoring study, it can be circumvented through the adjustment of class boundary cut off points of the EQR being applied to the data. The majority of stations in this study were classified as 'High' with none classified bellow the 'Moderate/Good' boundary. This meant that it was not necessary to adjust class boundary cut-off points to address the reduction of ES by subsampling sediments. Sub sampling sediments could also have an affect the recording of rare or less numerous species. In the case of standard habitat mapping studies, sub sampling sediments has been found to slightly affect sample affinity with macrofaunal distributions but not in an ecologically meaningful way (Somerfield and Clarke, 1997).

Our results indicate that there is potential in the INFOMAR data to be used in large scale predictive mapping exercises. The factors contributing to the marginally poorer performance of acoustic sediment models are mainly the result of local heterogeneity in sediment types within Galway Bay. The margin of difference in correct predictions between the acoustic sediment and observed sediment models is small, and arguably an acceptable margin of error when the potential cost savings associated with using readily available acoustic sediment classifications are taken into account. The asymmetrical distribution of samples across biotopes resulted in some being underrepresented in training datasets. This may also have affected model prediction rates (Hernandez et al., 2006). Confounding effects such as these can be expected to occur in heterogeneous habitats such as a large embayment containing islands and reefs. These effects are likely to be less pertinent in homogenous offshore habitats (Monteys et al., 2016, Brown and Blondel, 2009, Brown and Collier, 2008, Brown et al., 2011).

The methodology adopted for this study incorporated pre-existing datasets and cost effective field survey techniques. Our results indicate that this approach can be taken in other similar habitat mapping exercises. We have shown that INFOMAR acoustic classifications have potential to be used in large scale mapping studies such as those required under the MSFD. We have also presented a time effective technique for seafloor sediment data acquisition using expert judgement in the field. This methodology could be useful for rapid ground-truth studies for predictions made on sediment distributions (Stephens and Diesing, 2015) in new areas. Further research is needed to assess how INFOMAR acoustic classifications will perform in comparison to field sediment classifications in more homogeneous offshore habitats.

Regardless of sediment data acquisition methods, predictive spatial models will continue to play an important and growing role in the conservation of the ocean ecosystem (Dolan et al., 2008, Guinan et al., 2009a, Guinan et al., 2009b, Rengstorf et al., 2014, Rengstorf et al., 2013, Davies et al., 2008, Dutertre et al., 2013, Buhl-Mortensen et al., 2009), especially in relation to the sustainable management of marine resources (Council Directive 2008/56/EC). Co-ordinated seabed mapping initiatives such as INFOMAR are a useful asset for broad scale mapping exercises and Ireland is well equipped relative to many EU member states in this regard (Diesing et al., 2009). We have shown that it is possible to build a predictive model for EUNIS biotope distributions in a cost effective manner using available resources in combination with targeted data acquisition. The methodologies presented here have the potential to be applied in the offshore to meet Irelands' mapping obligations under the MSFD. Further research is needed to compare the effectiveness of INFOMAR versus observed sediment class based predictive models for the spatial distributions of EUNIS biotopes in offshore habitats.

3.4 Anthropogenic impacts on benthic communities

Case studies 1 and 2 assessed the impacts of intertidal oyster trestle cultivation on the benthic communities of intertidal sandflats that occurred in, or adjacent, to Natura 2000 sites. Organic enrichment is commonly associated with various modes of aquaculture (Borja et al., 2009b) including trestle cultivation (Nugues et al., 1996). In both studies we found no evidence of significant organic enrichment as a result of the deposition of faecal material in the sediment beneath trestles. Levels of organic matter in sediments beneath trestles were largely comparable to access and control treatment areas. The low levels of organic matter are most likely a result of the high tidal amplitudes that rinse the sediments twice daily, removing the finer mud particles and associated organic material. Similar studies have attributed similar low levels of organic matter in sediments at trestle cultivation sites to high dissipative nature of the sites (De Grave, 1998, Mallet et al., 2006). The drop in organic matter in sediment at sites after the stormy period of winter 2013/2014 suggests that further rinsing occurred during this period. By preventing the accumulation of organic material local hydrographic conditions are probably among the most important factors likely to influence interactions between oyster trestle cultivation and conservation features of intertidal habitats considered in Natura 2000 sites (Forde et al., 2015).

Macrofaunal community structure was highly variable in both Case Studies (1 and 2). The variability in macrofaunal distributions was not linked to organic matter levels or indicative of any significant impact at any one treatment area in either study. There was a notable increase in the proportion of small bodied first order opportunists in the 2014 study compared to the 2013 study after the intervening stormy period. This is viewed as a response of the opportunistic fauna to the recent disturbance caused by the winter storm events and was not associated with cultivation activities.

Case Study 1 also found no significant effect of treatment on biomass or secondary production suggesting that the quality of macroinvertebrate prey items available to foraging birds was not significantly affected by aquaculture activity across the Treatment areas. This result can be considered as an indirect indication that intertidal shellfish cultivation is not having an adverse effect on aspects of the conservation objectives of bird species (for which SPAs are designated) relating specifically to habitat suitability and food resources (Forde et al., 2015).

Despite using high levels of replication (n= ten replicate cores per Treatment area) it was not possible to detect any impact effects using standard univariate or multivariate analyses alone. Both studies used the IQI to overcome the potential confounding effect of highly variable community structure between treatment areas at each site. Critically, access routes across the sites were found to have a significantly higher number of samples that fall below the critical 'Moderate/ Good' boundary in both studies. Initially, this impact was thought to potentially be a temporary impact that would dissipate seasonally with the sediment resuspension (storm) events that occur in the NE Atlantic during the winter months. The intervening winter storms of 2013/ 2014 were significant and caused major infrastructural damage along the coasts of Ireland and the UK (Kendon and McCarthy, 2015). A disturbance event of this magnitude is representative of the highest levels of physical disturbance that can occur naturally in this region. The persistence of the access route effect despite this level of physical disturbance is indicative that this compaction effect could permanently fragment the benthic communities of intertidal sandflats within Natura 2000 sites around Ireland.

3. Discussion

The impacts associated with oyster trestle cultivation are arguably negligible when compared to other modes of aquaculture (Karakassis et al., 2000, Buschmann et al., 2006, Kalantzi and Karakassis, 2006, Giles, 2008, Borja et al., 2009b). Managing the fragmentation effect of access track compaction will require further research to develop an optimal approach. Periodic rotation of the access track may help to alleviate permanent fragmentation boundaries. Similarly, restricting the access route to one permanent track may be the best way to contain the risk posed to the conservation objectives of Natura 2000 sites.

Case Studies 3 and 5 examined the potential impacts associated with tidal energy extraction on the benthic environment and the potential risks that could be posed to the conservation objectives of the Strangford Lough Natura 2000 site. The epifaunal community of the Narrows was found to be spatially variable, with most of the variability being a result of localised differences in the abundances of the faunal dominants. The faunal dominants in both studies were the hydroids *Sertularia argentia* and *Tubularia indivisa* and the sponges such as *Halichodria panicea, Amphilectus fucorum* and the morphological type group 'mixed faunal turf'.

Significant spatial and temporal variability could be detected within this highly variable community. In Case Study 3 an asymmetrical BACI design found significant temporal variability to equally affect all stations. This variance was associated with trends in natural variability within the Narrows. The impact site was positioned 20 m downstream of the south Eastern rotor and covered 5 m^2 of the seabed. Our models could detect significant change when it occurred, and the HEHS index responded to indicators of disturbance when they occurred. In order to definitively assess the effects of energy removal on the epifaunal reefs of the Narrows a high resolution assessment was carried out in April and May 2013 (Case Study 5).

In Case Study 5 the epifauna in the greater survey area appear resilient to any negative influence the SeaGen may be having on the physical environment outside of one rotor diameter from the SeaGen. A significant change in hydrodynamic conditions or sedimentation rates would be necessary to negatively affect benthic communities within this type of high energy environment (Neill et al., 2009, Shields et al., 2011, Miller et al., 2013). Our results suggest that in tidal rapids that contain single devices, these large scale impacts are unlikely to occur.

Dividing the survey area into three treatment areas allowed spatial pattern in community structure to be detected. Assessing community structure in a radial manner around the SeaGen showed that the established measures of sensitivity for this habitat are negatively influenced in a restricted area at the foot of the SeaGen. Treatment area D1 was found to have significantly higher average multivariate dispersions around group average Bray-Curtis similarities compared to the other treatment areas. All treatment areas had epifaunal communities that were significantly different to one another in terms of the relative abundances of dominant species. This suggests that there is a gradual change from significantly altered communities in D1 to ambient 'normal' communities at D3, with communities at D2 existing in an intermediary state.

Bare rock was more widely distributed within D1 than anywhere else. This indicates that ambient physical conditions are being influenced by the turbine in a way that is detrimental to resident epifauna, as bare rock is an indication of recent disturbance (Chapman, 2002a, Chapman, 2002b, Chapman, 2003, Chapman, 2007).

D1 is the only treatment area that has an average ES of Moderate. The "Moderate/Good" boundary is deemed the critical boundary under the WFD, with remedial management action required to restore areas classified as Moderate or worse, to Good or better. According to the HEHS index the SeaGen is having a negative influence on epifaunal communities immediately between and adjacent to the device legs. However, this effect appears to dissipate quickly with increased distance from the device. The HEHS index did not find epifaunal communities within D2 to be affected. This indicates that the intermediary state community in D2 detected by multivariate faunal analyses exists in the realm of natural variability for the habitat.

Given that installation occurred in 2008 and substrates in D1 were still not fully colonised in 2012, it is unlikely that there is a lasting construction effect. To definitively assess the recovery rate of this community, a control area of boulder reef that is spatially distant from the turbine would need to be totally denuded and the subsequent colonisation rates monitored (Chapman, 2002a, Chapman, 2007). It is most likely that substrate colonisation within the footprint of the turbine is being held at an early successional stage as a result of the presence of the turbine.

When compared to certain methods of aquaculture such as finfish cage culture and high density longline mussel culture, tidal turbines have minimal benthic impacts. Aquaculture methods such as finfish cage, net pen culture, and long-line mussel cultures can have an impact on benthic communities directly beneath and downstream of the culture sites (Karakassis et al., 2000, Buschmann et al., 2006, Kalantzi and Karakassis, 2006, Giles, 2008, Borja et al., 2009b). This study shows that tidal energy extraction has a negligible impact on the benthic environment. Others have found the benthic effects of marine renewable energy devices to be beneficial to ambient communities (Reubens et al., 2013, Bergström et al., 2013, Langhamer and Wilhelmsson, 2009). Given the knowledge and 'acceptability' of the impacts of aquaculture, there is real potential for the significant growth in the production of marine renewable energy at minimal expense to the immediate environment. The localised influence of the turbine is a best case scenario when considering possible negative effects proliferating from a manmade instalment within a conservation site.

The installation of multiple devices in a small area could affect benthic communities in a significant manner. Issues such as the interaction of multiple turbulent wakes and the interaction of intermediate communities around the devices in a small area will require further investigation.

The ES of the subtidal sedimentary habitats included in Case Study 6 of Galway Bay is unaffected by the putative sources of anthropogenic impacts within the inner bay. This is largely due to all stations surveyed being at a significant distance from the sewage treatment outfall pipe, the dredge disposal ground and the Galway docks (O'Reilly et al., 2016, Patterson et al., 2006). All of these putative disturbance sources occur in depths less than 10 m and up into the intertidal, whereas our stations lay bellow 10 m.

3.5 Conclusion

The case studies in this thesis outline the potential that benthic EQRs have for monitoring a range of anthropogenic activities in fundamentally different habitats (Forde et al., 2015, O'Carroll et al., 2016, O'Carroll et al., 2017a, O'Carroll et al., 2017b in review, O'Carroll et al., 2017c in review). The application of WFD EQRs in new habitats further strengthens their ability to integrate benthic monitoring across the different Directives. Benthic EQRs also provide an effective way to accurately assess the spatial extent of anthropogenic impacts in ways that standard analyses cannot (Forde et al., 2015, O'Carroll et al., 2016). The ability of benthic EQRs to detect impacts that may be missed by univariate diversity indices or multivariate community analyses, and their amenability to modification, makes them useful tools capable of strengthening the conservation goals of the Directives. Prior to the application of a benthic EQR, adequate consideration must be made on the habitat and impact source in question, and the EQR being used. The lack of sufficient planning or calibration studies could result in erroneous and costly remedial management decisions being made (Quintino et al., 2006).

Hydrodynamic model outputs were not found to provide any explanation of spatial variance in benthic community structure in the case studies presented in this thesis (Kregting et al., 2016, O'Carroll et al., 2017b in review, O'Carroll et al., 2017c in prep.). When the studies presented here are compared with those that have found correlations between community structure and hydrodynamic parameters there is one key difference. Our survey areas are relatively homogenous when compared to those of others (Dutertre et al., 2013, Christensen et al., 2009). This homogeneity in community structure may account for the apparent lack of relationship between hydrodynamic model outputs and community structure in our case study. In future studies intending to use hydrodynamic model outputs we recommend that bathymetric data be incorporated into the model. We also recommend that serious consideration be taken prior to commitment to invest in hydrodynamic model development. If current models are most useful for discerning between starkly different bottom communities such as sediment from cobble, boulders or reef, this capability is already present in acoustic bathymetric data (Brown et al., 2011).

3. Discussion

Other studies have previously been shown that the reduction of taxonomic effort in monitoring exercises does not significantly affect outputs (Kennedy et al., 2011). In this thesis we also demonstrate that this may also be true when monitoring in tidal energy extraction sites (O'Carroll et al., 2017b in review). This could be taken into account during survey planning stages to reduce costs and save time. We also demonstrate that it is possible to build an effective predictive modelling approach using publically available data containing multiple confounding effects along with targeted data acquisition in a cost effective manner (O'Carroll et al., 2017 in prep.). Efficient survey techniques such as sediment classification by expert observation in the field, and sediment sub sampling grab samples from sediments appear to be acceptable operating procedures in large scale top down mapping exercises. The INFOMAR dataset has the potential to be used in offshore mapping exercises as a surrogate for standard sediment classifications, but further research is required to test this.

Oyster trestle cultivation and tidal energy extraction do not appear to have significant deleterious impacts on the designated waterbodies in which they are occurring (Forde et al., 2015, O'Carroll et al., 2016, O'Carroll et al 2017a, O'Carroll et al., 2017b in review). The access track compaction effect is an unavoidable phenomenon, yet there may be an optimal way to manage this impact over time. The benthic footprint of a tidal energy turbine does not extend outside the immediate vicinity of the device mooring structure and is arguably negligible. In both instances, especially in the case of tidal energy turbines, the associated impacts on the benthos are as small, and do not appear to pose a significant risk to the conservation status of the sites in which they are operational. When considering potential negative effects proliferating from an anthropogenic activity, both intertidal bivalve trestle cultivation and tidal energy extraction should be considered as environmentally sustainable and socioeconomically beneficial.

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Article Chapter I

Impact of intertidal oyster trestle cultivation on the Ecological Status of benthic habitats

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James Forde, Francis X. O'Beirn, Jack P.J. O'Carroll, Adrian Patterson, Robert Kennedy

Article Chapter II

Impact of prolonged storm activity on the Ecological Status of intertidal benthic habitats within oyster (Crassostrea gigas) trestle cultivation sites

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Identifying Relevant Scales of Variability for Monitoring Epifaunal Reef Communities at a Tidal Energy Extraction Site

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Do Changes in Current Flow as a Result of Arrays of Tidal Turbines Have an Effect on Benthic Communities?

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RESEARCH ARTICLE

Do Changes in Current Flow as a Result of Arrays of Tidal Turbines Have an Effect on Benthic Communities?

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Abstract

Arrays of tidal energy converters have the potential to provide clean renewable energy for future generations. Benthic communities may, however, be affected by changes in current speeds resulting from arrays of tidal converters located in areas characterised by strong currents. Current speed, together with bottom type and depth, strongly influence benthic community distributions; however the interaction of these factors in controlling benthic dynamics in high energy environments is poorly understood. The Strangford Lough Narrows, the location of SeaGen, the world's first single full-scale, grid-compliant tidal energy extractor, is characterised by spatially heterogenous high current flows. A hydrodynamic model was used to select a range of benthic community study sites that had median flow velocities between 1.5–2.4 m/s in a depth range of 25–30 m. 25 sites were sampled for macrobenthic community structure using drop down video survey to test the sensitivity of the distribution of benthic communities to changes in the flow field. A diverse range of species were recorded which were consistent with those for high current flow environments and corresponding to very tide-swept faunal communities in the EUNIS classification. However, over the velocity range investigated, no changes in benthic communities were observed. This suggested that the high physical disturbance associated with the high current flows in the Strangford Narrows reflected the opportunistic nature of the benthic species present with individuals being continuously and randomly affected by turbulent forces and physical damage. It is concluded that during operation, the removal of energy by marine tidal energy arrays in the far-field is unlikely to have a significant effect on benthic communities in high flow environments. The results are of major significance to developers and regulators in the tidal energy industry when considering the environmental impacts for site licences.

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Introduction

Renewable energy from wave and tidal technology has the potential to contribute significantly to energy security for future generations. Realistic estimates of the commercial potential of the two forms of marine energy indicate there is the possibility of up to 337GW being installed worldwide by 2050 [1]. Although the commercial development of the technology is still in the early stages, it appears that the potential exploitation of tidal energy is significantly closer to realisation than that of wave energy [2]. This reflects to some extent the higher predictability of the tidal resources on a daily basis compared to wave energy and other renewable energy resources such as solar and wind power.

Common to both tidal and wave energy extraction systems is concern regarding the potential environmental consequences of the deployment of the technology. The environmental concerns reflect differences in both the form of interaction of the system technologies with the environment and in the overall dynamics of the areas suitable for exploitation (e.g. [3, 4]). In the case of tidal energy converters (TEC), the majority either planned or constructed are characterised by large underwater rotors together with substantial supporting structures or mooring systems. Large arrays of these structures have the potential to induce significant changes in the hydrodynamic field [5]. It has been suggested that the altered hydrodynamics can have both near- (immediately around the structure) and far-field (wake and basin scale) effects which include a reduction in mean flow, tidal flow diversion, increased bottom drag and changes in tidal phasing and height [6, 7, 8]. The presence of TECs and the changed hydrodynamic field may result in direct and indirect effects on local populations of diving seabirds, fish and mammals and also, in the case of the hydrodynamics, on the benthos [3, 9, 10].

Owing to the requirement for high (>2 m/s) flow velocities, the location of TEC will be restricted to areas where seabeds are characterised by rocks, boulders, cobbles and sand [11]. Although the spatial extent of such areas is limited, the benthic communities associated with these locations are relatively pristine as they have undergone minimal exploitation or disturbance by anthropogenic influences owing to their exposed nature. To date the majority of surveys of these areas have been spatially restricted and generally undertaken for broad conservational objectives [11]. However, given present moves towards the commercial development of tidal energy, it is necessary to have a close understanding of the interplay between the physical dynamics and the biology of representative organisms in the control of benthic community structure in highly dynamic areas. Clear links have been observed between the spatial patterns for marine communities with substrate type as a direct causal result of modelled or measured current speed in which communities live (e.g. [12, 13, 14, 15]). Therefore the outcomes of investigations linking the hydrodynamics with the biology have the potential to allow predictions to be made of changes in benthic communities resulting from flow changes induced by TECs.

Preliminary investigations of the effect of a single, full-scale TEC, SeaGen, on the benthos showed somewhat surprisingly minimal influence of the device on the seabed communities [16]. Of greater importance, though, is the potential impact of arrays of TECs on the benthos. The number of turbines required to extract a significant amount of the energy is of the order of 10s to 100s. Such arrays are likely to extract much larger amounts of energy from the ambient environment resulting in potentially measurable reduction in flow velocity; however, the general lack of understanding of the interaction between arrays and the marine environment has been identified as one of the four major factors impeding the development of tidal energy as a commercial resource [17]. Advances in the application of high resolution flow models are allowing the detailed prediction of the current regime in highly dynamic areas [18] with these, in turn, allowing the design of more effective benthic surveys in the challenging conditions typifying these locations and allowing opportunity for more detailed analysis of relationships.

The aim of this study was to determine whether any observed spatial variation of macrobenthic communities could be related to changes in a natural velocity gradient in a narrow channel dominated by strong semi-diurnal tides. The approach taken was to obtain macrobenthic community data using a drop down video survey system at a number of sites in the channel spanning a range of ambient current velocities relevant to the tidal energy industry to test the null hypothesis that there was no significant difference in community structure over the gradient of flow velocities sampled. Without a tidal array in existence, this is presently the best approach in predicting the influence of TEC arrays on benthic populations located within the area of an array. This would assist regulators and the tidal site developers in site selection and environmental licensing.

Methods

Sampling site selection

Sampling was carried out in the Strangford Narrows in Northern Ireland, the location of SeaGen, the world's first single full-scale, grid-compliant tidal energy extractor. It is representative of a site with relevant flow fields to the tidal energy industry [16] (Fig 1). The velocity range over which samples were taken was dictated by prediction of the operational flow velocities of a TEC below which the installation of a tidal turbine would be considered commercially unfavourable. To establish the range, it was first necessary to determine the velocity changes that would result from the presence of an array of TECs. It is to note here that this work was not based on the SeaGen design but rather the type of device most likely proposed for commercial development, the horizontal axis tidal turbine (HATT) [19]. Two ways to quantify this change were employed: assessment of (i) the effect of a single or array of TECs on flow velocity and (ii) the resource limit at which commercial exploitation is not viable, i.e. the point at which a commercial company would not add further turbines into a location.

When considering the potential effect of arrays of TECs on the mean flow, it is important to note that such turbines are unlikely to extract more than 50% power from the rotor swept area (the theoretical Betz Limit is 59.3%, though this is not achieved in real world applications [20]. Since hydrokinetic power is proportional to the velocity cubed, a 50% reduction in hydrokinetic power will only result in a 20% reduction in flow velocity. Further, this reduction only applies to the rotor-swept area of a device which, for present TECs, covers only a small area of the water column (the diameter of the rotor-swept area) due to technical, navigational and environmental considerations. Thus relatively large changes in power extraction by TECs will be reflected in only small changes in near-field flow speeds. Drag losses associated with the support structure of the TEC will also extract energy from the flow and assist in reducing flow velocities, although the turbulent nature of the marine environment results in fast recovery of the ambient velocity field: Savidge et al. [16] reported velocities close to the original undisturbed currents ($>$ 95% of undisturbed velocity) at 5 rotor diameter downstream (80 m). Further it was not observed that the infrastructure of SeaGen increased velocity or turbulence close to the seabed [21]. The dynamics of wake structure and recovery in relation to TECS are poorly understood with absolute predictions likely to be device-specific. In addition, use of modelling the wake velocity as an estimator for the impact of TECs on benthic habitat is considered premature as it is not yet computationally possible to numerically model at the individual device scale to resolve blade and pile wake turbulence as well as span far-field to determine changes in hydrodynamics as a result of tidal arrays [5].

The second approach to defining a velocity range relevant to benthic interactions associated with the installation of TECs is to assess the resource limit beyond which commercial exploitation is not viable. As noted above, although tidal energy converters are still in early

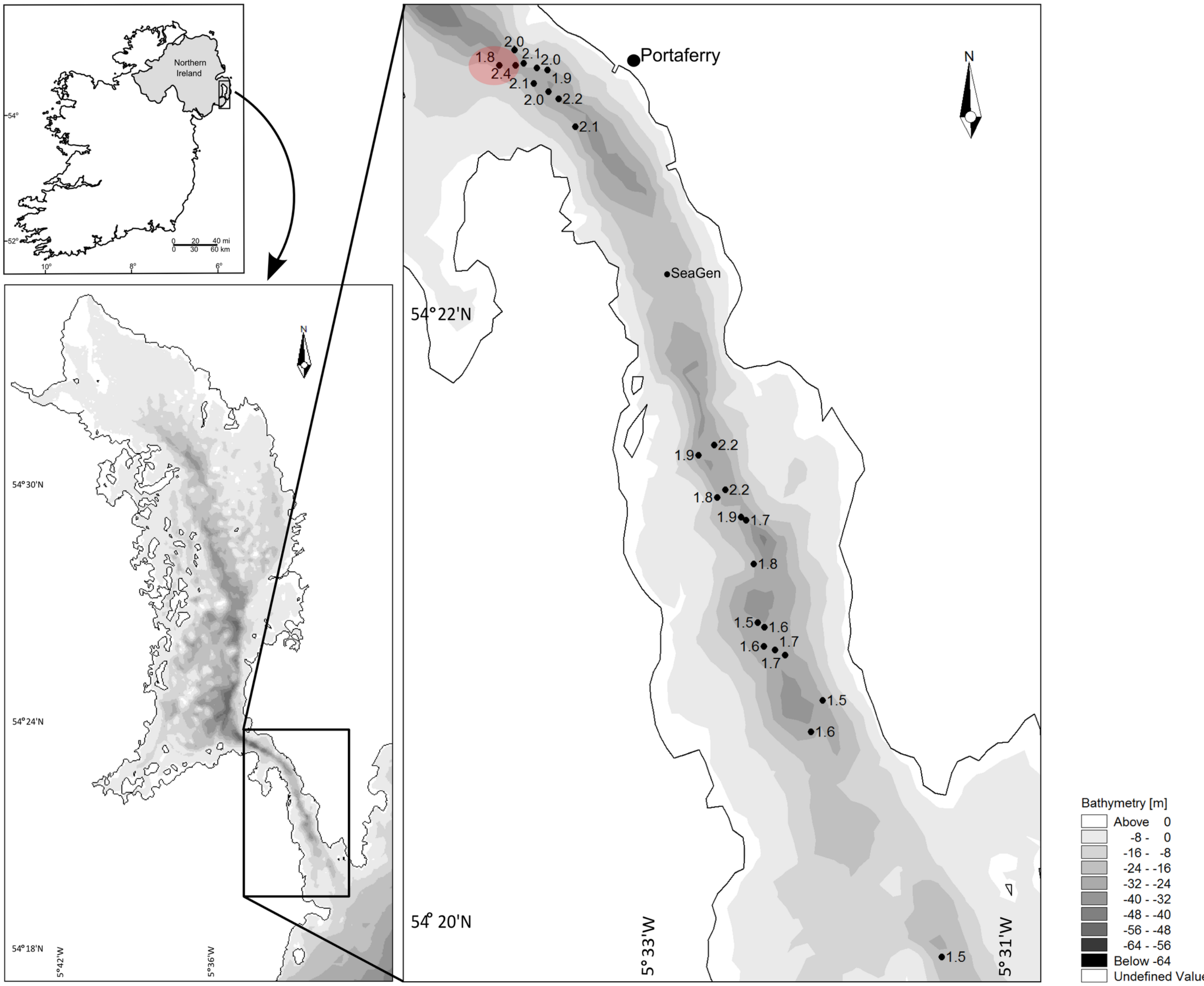

Fig 1. Site locations with P50 value (median velocity) within Strangford Lough Narrows. Red circle highlighting the heterogeneity of flow over a distance as little as 100 m.

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development, the HATT is presently the type of device most frequently proposed for commercial development [19]. HATT have characteristic cut-in speeds, i.e. velocities below which they do not function, and have design or rated speeds at which the device obtains full power output. For example the tidal turbine SeaGen at Strangford Lough has a cut-in velocity of 0.8 m/s and a rated velocity of 2.5 m/s.

Table 1 provides the percentage utilisation (power available in the flow field in relation to the power derived from the rated speed of a device) based on three hypothetical cut-in and rated velocities for three different hypothetical turbines: A, B & C. It should be noted that this is only the power available in the flow field: considerations of energy conversion efficiency to mechanical power and drive train losses are excluded here, as again they are device specific. Although no relevant data are publically available and comparable figures for the wind industry

Table 1. Resource requirements for three different hypothetical tidal energy converters (A, B & C) and percentage power availability for tidal energy generation (i.e. the power available to a turbine in real tidal flow (P50, median velocity) in relation to the hydrokinetic power obtained from the rated velocity).

		Resource of P50 (median) velocity		
	cut in / rated velocity of turbine (m/s)	1.5 m/s	2 m/s	2.4 m/s
Turbine A	0.6 / 2.2	36%	63%	74%
Turbine B	0.8 / 2.5	24%	52%	67%
Turbine C	1.2 / 2.8	17%	40%	58%

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are not representative, information obtained by the authors from collaborations with various turbine developers indicate that a realistic percent utilisation for a feasible project is approximately 60% and may drop to just above 40%. This latter figure is taken as a clear minimum as it is unlikely even for a mature technology to go below this limit due to the cost of the technology, including economy of scale considerations [17]. From Table 1 it can be seen that, presently, median flow velocities of 2 m/s need to be reached or exceeded to meet the velocity requirements of current tidal energy devices. Therefore, to incorporate the predicted operating velocities of the developing current tidal turbines and associated limits, median current values ranging from 1.5 to 2.4 m/s were chosen for the study. It is to be noted that the lower limit of 1.5 m/s defines a high current environment.

Use of the median velocity (velocity that is exceeded 50% of the time (P50)) is not commonly employed in the characterisation of high current environments; more usually peak mean neap and spring tidal current velocity values are used to define these environments. Table 2 compares three different median velocities of 1.5, 2.0 and 2.4 m/s and the approximate associated peak tidal current speeds for typical (mean) neap and spring conditions for the Strangford Narrows. From this table, it is apparent that only the neap peak flows and the median flows are similar. In contrast to peak current flow values, the median gives a measure of how often the defined flow occurs and hence is a more meaningful parameter for this study compared to the conventional peak or mean flow values.

Based on the above considerations, the locations of the sites to be used for sampling of the benthos were selected from the output of the Strangford Lough current model based on MIKE 21 modelling software (DHI Water and Environment software package; www.dhisoftware.com) [18]. The hydrodynamic model has a flexible mesh and uses a cell-centred finite volume method to determine the current field by solving a depth averaged shallow water approximation (full details of the development and calibration of the model can be found in [18]). Cell size averaged approximately 50 m resulting in a total of 52,882 cells throughout the domain. The model was run over a three month period (February—April 2011) incorporating a range of neap and spring tide conditions. The outputs were recorded as Reynolds averaged velocities over 5 minute intervals; these values exclude flow fluctuation due to turbulence. The model does take into account benthic boundary layer processes even though depth averaged velocities

Table 2. Relationship between P50 (median velocity) and typical peak neap and spring velocities for three hypothetical sites (A, B & C) in the Strangford Narrows

	Site A	Site B	Site C
P50 (median) velocity (m/s)	1.5	2.0	2.4
Mean neap tidal velocity (m/s)	1.5	2.0	2.4
Mean spring tidal velocity (m/s)	2.2	3.0	3.6

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are taken:

$$\frac{\bar{u}}{u\star} = \frac{1}{\kappa} \ln\left(\frac{h}{eZ_0}\right) \quad (1)$$

where $\bar{u}$ is the depth averaged velocity, $u_\star$ is the shear velocity, κ is the von Kármán constant (0.4), h is the water depth, e is the Eulerian number and Z_0 is the friction height. It describes the relationship between $\bar{u}$ and $u_\star$ of a fully developed boundary layer. Based on a Z_0 in the order of 0.1–0.5 m [16], the Reynolds number can be assumed to be always $> 10^5$ for median velocities. This means a fully turbulent boundary layer is to be expected in the study area. While Z_0 may vary from location to location, in general the bed in the Narrows is rough with boulders and bedrock. Thus a direct relationship between the depth averaged and the shear velocity exists, which influences the forces on the fauna and flora at the seabed.

Three replicates of each of nine defined P50 values (1.5, 1.6, 1.7, 1.8, 1.9, 2.0, 2.1, 2.2, 2.4 m/s) were randomly selected from within the Narrows except for the highest velocity, 2.4 m/s, for which only one replicate was available; no sites with P50 values of 2.3 m/s were able to be established (Fig 1). In total, 25 sites (cells) were sampled. While SeaGen has been shown to have minimal influence on seabed communities [16], cells within the 1 km radius around SeaGen were excluded from site selection in order to reduce the possibility of local bias. To control for potential effects of depth on the structure of the benthic communities, all sampling locations were restricted to a depth range of 25–30 m.

Sampling

The 25 designated sites were sampled between 25 April and 23 May 2013 at slack water for substrate type and macrobenthic community using a drop down video survey method. A video quadrat camera system was constructed from a stainless steel frame with a GO-PRO HERO3 Black Edition (San Mateo, USA) mounted 0.4 m above the substrate to provide a quadrat size area of 0.5 × 0.5 m. Light was provided by three underwater torches (Kinetics Mini Q40 eLED Plus) placed strategically on the frame to provide an even light source across the substrate. At each site the frame was gently lowered to the substratum with the video continually recording (1080p/48 FPS with medium field of view) for approximately 5–10 seconds then lifted, moved ~ 2 m and dropped to provide 30 random quadrats within an approximate 20 m^2 area starting at the centre coordinate of the mesh cell from which the velocities were extracted as determined from the model. Visual of the seabed during the video survey allowed for an estimate of the distance between drops. Due to tidal constraints, only one site was sampled each slack tide. Field studies in this area did not require any permit or permission and did not involve any protected or endangered species.

Image and data analyses

The 21 best resolved still images from the 30 quadrats (630 images in total) were grabbed from the video footage using Windows Media Player. Image selection was based on clarity of the focused frame and for consistency, only one skilled marine benthic taxonomist observed the images creating a species list for the analysed frame after three separate viewings. Viewings took place with a 48 h period between sittings to avoid observer fatigue. The images were assessed for percentage cover and density of discrete individual species (identified to the nearest taxonomic level) based on a random point quadrat (100 points) methodology. Motile fauna including fish and decapods were removed from the data matrix before analysis as is typical for reef epifaunal studies [22]. The substratum composition for each quadrat was visually determined using the Joint Nature Conservation Committee (JNCC) biotope coding protocol [23].

A core assumption of the analysis is that attached sessile epifaunal distributions reflect the integration of the environmental gradients in the observational area over time. The percentage cover data and abundance were transformed to the ordinal Marine Nature Conservation Review (MNCR) SACFOR scale using the method of Connor et al. [11]. The SACFOR data were converted to a Bray-Curtis similarity matrix [24]. The null hypothesis of no significant difference in community structure was tested using a nested mixed model permutational analysis of variance (Permanova; [25, 26] in the Permanova+ package in Primer 6. The factor Site (the 20 m × 20 m area sampled on a particular tidal cycle) was random and nested within the fixed factor Velocity (the ordinal level of median velocity predicted for that sampling site in the hydrodynamic model). Species characterising the different levels of velocity were identified using the exploratory data analysis Simper [27, 28].

Results

A spatially heterogeneous flow can be observed in the Narrows with variation in flow speeds of almost 0.5 m/s over distances as small as 50–100 m (see Fig 1). As expected, greatest flow velocities were observed in the narrowest part of the channel, near Portaferry, with velocities ranging from 1.8 to 2.4 m/s in this area. Lowest velocities were observed towards the entrance of the Narrows where the channel is at its widest (Fig 1).

A diverse faunal and limited floral assemblage was observed throughout the Strangford Narrows with a total of 44 taxa recorded (Table 3; Fig 2). The main groups represented were cnidarians, molluscs and bryozoans, with crustaceans, sponges and echinoderms (Table 3). The faunal community of the study area was characterised by the presence of the sponge *Halichondria panicea* and other species of the Phylum Porifera, the polychaete *Spirobranchus* sp. and various members of the Phyla Bryozoa and Cnidaria. Other commonly occurring individual epifauna included the cnidarians *Sagartia* spp., *Alcyonium digitatum*, and *Obelia* spp., as well as representatives of the genus *Balanus* (Fig 2). Several mobile species were also present at $>$ 85% of the sites including the velvet swimming crab *Necora puber* and sea urchin *Echinus esculentus* (Fig 2A & 2D). A range of substratum compositions which matched JNCC recognised biotopes, was recorded throughout the Narrows from cobbles, through small boulder to large boulder all on a bedrock base layer.

Simper analysis (Table 4) demonstrated that all communities sampled corresponded to the EUNIS biotope CR.HCR.FaT (Very tide-swept faunal communities on circalittoral rock). Permanova analysis (Table 5) showed that when sampling site was nested as a random factor within velocity class, there was no significant effect of the P50 value on macrofaunal community structure. Thus within the narrow channel at the entrance to Strangford Lough, the composition of the epifaunal communities did not differ significantly within the median velocity range of 1.5 to 2.4 m/s, as defined by the P50 value.

Discussion

Most benthic ecological studies relating to the effects of anthropogenic disturbance have focussed on the consequences of adding energy to the seafloor in the form of dredging, fisheries-associated abrasion and spoil disposal. In contrast the present study took a reverse approach by focussing on the effects of removing energy from the environment. However few of the studies that have been carried out to establish relationships between current strength and biological distributions have been based on estimates of absolute current speeds, as opposed to qualitative estimates [29, 30]. Two exceptions to the qualitative approach are the studies of Ordines et al. [14] and Dutertre et al. [31] who investigated variance in benthic community distribution and substrate type. The effects of varying mean current flow on the

Table 3. Species list and abundances using the JNCC's (MNCR) SACFOR abundance scales for each of the 25 sites surveyed by drop down video camera between 25 April and 23 May 2013. S = Superabundant, A = Abundant, C = Common, F = Frequent, O = Occasional, R = Rare.

P50 velocities (m/s)	**1.5**			**1.6**			**1.7**			**1.8**			**1.9**			**2.0**			**2.1**			**2.2**			**2.4**
Replicates	**1**	**2**	**3**	**1**	**2**	**3**	**1**	**2**	**3**	**1**	**2**	**3**	**1**	**2**	**3**	**1**	**2**	**3**	**1**	**2**	**3**	**1**	**2**	**3**	**1**
Phylum																									
Annelida																									
Spirobranchus	R	O	O	R	O	O	O	O	O	O	O	R	O	O	R	O	O	R	O	O	O	O	O	O	O
Spirorbis (Spirorbis) spirorbis	-	-	R	-	-	R	R	-	-	R	-	-	-	R	-	-	-	-	-	-	-	R	-	R	-
Arthropoda																									
Cancer pagurus	-	R	-	-	R	R	R	R	R	R	R	-	O	R	R	-	R	O	O	R	O	R	R	R	R
Necora puber	-	R	-	-	R	R	R	R	R	R	R	R	-	-	R	-	R	R	R	R	R	R	-	R	R
Paguroidea	-	-	-	-	-	-	-	R	R	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Leptostraca	-	R	-	-	-	-	-	-	R	-	R	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Balanus	R	R	R	R	R	R	O	R	O	R	R	R	R	R	R	R	O	R	O	R	R	F	R	F	R
Bryozoa																									
Flustra foliacea	-	-	F	R	-	-	R	R	-	-	-	R	-	O	-	-	-	R	-	R	-	R	-	-	-
Alcyonidium	O	R	-	R	O	R	R	-	-	R	R	-	O	-	-	-	O	-	-	-	-	R	O	O	O
Alcyonidium diaphanum	R	O	R	O	O	O	R	R	F	R	O	O	O	F	O	R	R	R	O	-	-	O	R	-	O
Bryozoa	O	O	R	O	O	O	O	O	-	R	R	R	O	O	R	O	O	R	O	O	O	O	F	O	O
Chordata																									
Dendrodoa grossularia	-	R	-	-	-	-	-	-	-	-	-	R	-	-	R	R	-	R	-	-	-	R	-	-	-
Cnidaria																									
Urticina felina	-	-	R	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Capnea sanguinea	-	-	R	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Sagartia	R	R	R	O	O	R	R	O	R	R	R	R	R	-	R	R	R	R	R	O	-	R	O	R	R
Actiniaria	-	-	R	-	-	R	-	-	R	-	R	-	R	-	-	-	-	-	-	-	-	-	R	R	-
Alcyonium digitatum	R	C	R	F	F	R	O	F	C	O	C	O	O	F	R	O	R	R	F	O	R	F	O	O	R
Obelia	-	O	R	-	R	F	R	R	C	O	R	O	C	O	C	R	-	O	O	R	-	O	C	F	C
Halecium halecinum	-	-	-	-	-	R	-	-	-	-	-	-	-	O	R	-	-	R	-	-	-	-	-	R	-
Nemertesia	-	-	-	R	-	R	-	-	F	-	-	-	-	-	-	-	-	O	R	R	F	-	R	R	-
Nemertesia antennina	-	-	-	-	-	-	R	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Abietinaria	-	-	-	-	-	R	-	-	-	-	-	-	O	-	-	-	-	-	-	-	-	-	O	R	R
Hydrallmania	-	R	-	-	-	-	-	-	-	R	-	-	-	-	-	-	R	-	-	-	-	R	-	-	-
Hydrozoa	O	O	R	R	O	O	R	R	F	R	O	R	F	O	R	R	O	F	O	F	O	R	F	O	O
Echinodermata																									
Asterias rubens	-	R	O	-	R	-	-	-	R	-	R	-	-	R	R	-	R	R	-	-	-	-	-	R	R
Henricia oculata	-	-	-	R	R	-	-	-	-	-	-	-	-	R	-	-	-	-	-	-	-	-	-	-	-
Crossaster papposus	-	-	-	-	R	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Echinus esculentus	-	F	-	R	O	R	R	-	O	R	O	R	R	R	R	R	-	R	R	R	-	R	R	R	R
Mollusca																									
Mytilus edulis	-	-	-	-	-	R	-	-	-	-	-	-	R	-	-	-	-	-	-	-	-	-	-	R	R
Venus casina	-	-	R	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Turritella communis	-	-	-	-	-	-	-	-	-	-	-	R	-	-	-	-	-	-	-	-	R	-	-	-	-
Buccinum undatum	-	-	-	R	R	R	O	R	R	-	R	R	R	-	R	-	-	R	-	R	-	-	R	R	R
Nucella lapillus	F	-	R	-	R	R	R	R	R	-	R	R	R	R	R	-	C	R	R	O	O	-	R	R	R
Calliostoma zizyphinum	R	-	R	-	-	-	-	-	-	-	-	-	-	-	R	-	-	R	R	-	R	-	-	-	R
Gibbula cineraria	-	-	R	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Ochrophyta																									
Laminaria hyperborea	-	-	-	-	-	-	-	-	-	-	-	-	-	R	R	-	-	-	-	-	-	-	-	-	-

(*Continued*)

Table 3. (*Continued*)

P50 velocities (m/s)	1.5			1.6			1.7			1.8			1.9			2.0			2.1			2.2			2.4
Replicates	1	2	3	1	2	3	1	2	3	1	2	3	1	2	3	1	2	3	1	2	3	1	2	3	1
Porifera																									
Axinella	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	R	-	-	-	-	-	-	-	-	-
Halichondria (Halichondria) bowerbanki	-	-	-	-	-	-	R	-	-	-	-	-	-	-	-	R	-	-	R	-	-	-	-	-	-
Halichondria (Halichondria) panicea	O	F	-	F	F	O	F	F	C	F	F	F	F	F	F	F	O	C	C	F	R	F	F	O	F
Porifera	R	F	R	R	O	O	O	R	F	R	F	O	F	O	R	O	R	F	-	O	O	R	F	O	O
Rhodophyta																									
Phyllophora crispa	-	-	R	-	-	-	-	-	-	-	-	R	-	-	-	-	-	R	-	-	-	-	-	-	-
Palmaria palmata	-	-	-	-	-	-	-	-	-	-	-	-	-	-	R	-	-	-	-	-	-	-	-	-	-
Encrusting Coralline Algae	-	R	R	-	-	-	R	R	-	R	-	R	-	R	R	R	-	R	-	-	-	R	-	-	-
Rhodophyta	-	-	R	-	-	-	-	-	-	-	-	-	-	-	-	-	R	-	-	-	-	-	-	-	-

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distribution of suspension feeding epifauna such as corals [29] and *Mytilus* beds and sponge dominated communities on vertical rock walls [30] have also been assessed, with it being shown that flow velocities significantly influence the distribution of these suspension feeding communities. A clear link is apparent between the spatial patterns observed for marine communities and substrate type over a wide range of ambient velocities. This conclusion is based on the assumption that substrate is controlled by local ambient velocity (e.g. [12, 13, 14, 15]). In the present study, substrate type was relatively homogenous and no changes in benthic communities at the high end of the velocity spectrum over almost 1 m/s velocity range were observed, this most likely reflecting the highly physically disturbed nature of the environment.

High velocities > 1.5 m/s are characteristic of flow rates found in a number of locations around the UK, with several of these locations being actively sought for exploitation by the tidal energy industry. Habitats in these environments have been described as rich in terms of biodiversity and production [11, 32]. However in comparison to low energy flow environments, here defined as < 1 m/s, few studies have been carried out that quantitatively describe the natural variability of the benthic communities in these high energy flow environments [15, 33, 34] and none directly related to absolute velocities. This is perhaps unsurprising given that marine soft-sediment habitats, associated with low velocity environments, are the most common benthic habitats in the marine environment covering approximately 70% of the planet [35]. The logistical difficulties in sampling high flow environments due to the reduced time available for sampling and limitations in sampling methods [36] will also have contributed to the general lack of detailed investigation of these areas. For example, SCUBA sampling is costly and constrained by the tidal velocity and depth, while grab samples cannot be used on hard substrata such as rock and boulders. Despite the limitations in sampling in these physically challenging environments that may also miss, for example, certain localised fauna such as crevice species, reef systems in these environments have been classified qualitatively as rich in terms of biodiversity and secondary production [11] and can be considered ecologically important environments. In addition, the majority of these high energy environments have been subject to minimal anthropogenic influence.

The species observed in the present study were consistent with those typically recorded for high current flow environments from other comparable benthic studies around the UK [11, 33, 34]. The most common species recorded in the Strangford Narrows were the soft coral deadmen's fingers *Alcyonium digitatum*, the sponge *Halichondria panacea*, and various members from the Bryozoa, Cnidaria and Porifera Phyla most noticeably *Alcyonidium* spp., *Obelia* spp., *Sagartia* spp., and representatives from the spp. *Hydrozoa* and *Bryozoa* spp. (Table 4). The

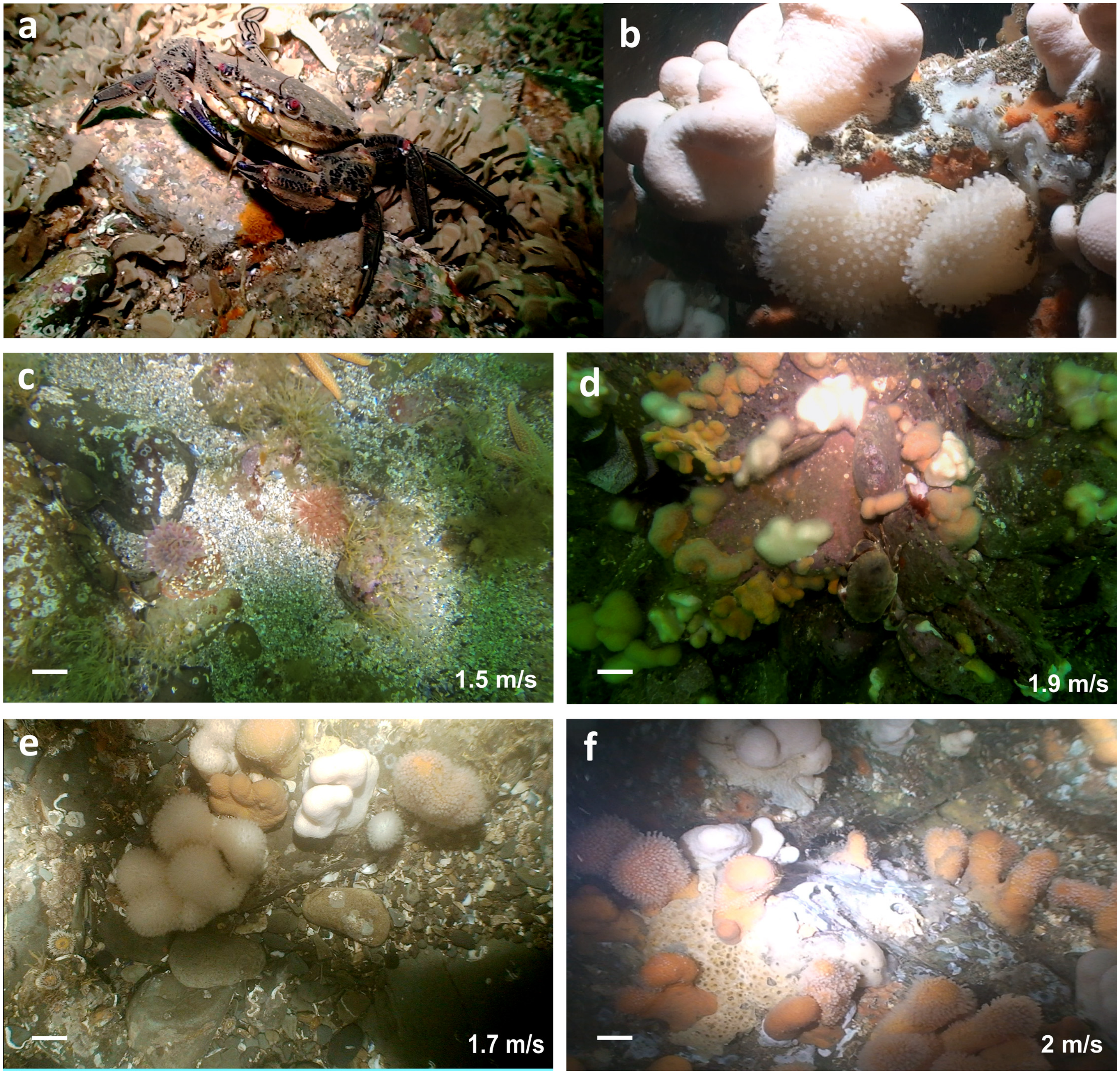

Fig 2. Anemones, soft corals, crustaceans and sponges inhabiting the seafloor of the Strangford Narrows, Strangford Lough. Close up photos of the velvet swimming crab *Necora puber* (a) and dead man's fingers *Alcyonium digitatum* (b); benthic quadrat images derived from the video footage that were used for the analysis from the sites with flow rates 1.5, 1.7, 1.9 and 2 m/s (c-f). Scale bars represent approximately 0.05 m.

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communities correspond to very tide-swept faunal communities in the EUNIS classification [11, 33, 34]. The present investigation provides valuable insight into the diversity that is present in the Strangford Lough Narrows, but which may also be expected in other comparable areas where resource consent is being sought for tidal energy investment.

Table 4. Simper analysis of taxa characterising the sessile epifaunal community of Strangford Lough Narrows.

Species	Av.Abund	Av.Sim	Sim/SD	Contrib%	Cum.%
Halichondria (Halichondria) panicea	2.3	9.13	1.21	23.18	23.18
Spirobranchus	0.96	6.13	1.55	15.56	38.74
Bryozoa	0.99	4.44	1.3	11.28	50.02
Alcyonium digitatum	1.57	4.08	0.61	10.36	60.38
Porifera	1.3	3.02	0.59	7.66	68.04
Hydrozoa	1.22	3.01	0.64	7.65	75.69
Obelia	1.36	2.83	0.47	7.19	82.88
Balanus	0.81	2.04	0.6	5.18	88.06
Alcyonidium diaphanum	0.98	1.56	0.37	3.96	92.02
Average Similarity 34.4%					

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The lack of change in the benthic epifauna community over a velocity range of almost 1 m/s presents a challenge for assessing possible effects of tidal energy devices on the local benthic communities in the context of high energy, physically disturbed and highly variable locations [16]. However, despite the apparent homogeneity in the distribution of the benthos recorded in the present study, the environment in the Strangford Narrows is by no means homogenous physically. Variation in flow speed is appreciable over distances as small as 50–100 m with flow speeds varying by almost 0.5 m/s as indicated in red on Fig 1. Overall the lack of a significant effect of current speed on the composition of the benthic communities or distribution of specific taxa suggests that the communities are adapted to the high physical disturbance associated with the strong current flows in the Strangford Narrows. Disturbed boulder habitats are characterised by faunal communities that cover a broad spectrum of successional states. Sousa [37, 38] described spatial community structure on boulder tops as a heterogeneous mosaic, with each boulder top as “a patch of habitat which differs in size and age from that of neighbouring boulders”. These communities therefore could be observed as a mosaic of opportunistic species and individuals being continuously and randomly affected by turbulent forces and physical damage from abrasion and contact with sand and larger sediment particles being transported along the seabed.

The results of this study are of major significance to developers and regulators in the tidal energy industry and may provide a model of the effects of energy removal on community type that can be used to determine when a development has had a significant effect on the local communities. Development of hydrodynamic models predicting the flow perturbations associated with the deployment of tidal energy devices will allow the regulators to relate predicted changes in absolute velocities obtained as output from models to the effects on benthic communities. We also wish to emphasise that the benthic data obtained in the present study is of a form that is typically available to environmental managers when assessing proposed

Table 5. Permanova analysis of Bray Curtis similarity matrix derived from epifaunal distributions in Strangford Lough Narrows measured on a SACFOR scale.

						Unique	
Source	**df**	**SS**	**MS**	**Pseudo-F**	**P(perm)**	**perms**	**P(MC)**
Velocity	8	1.08E+05	13457	0.89696	0.6637	9881	0.6678
Site (Velocity)	16	2.40E+05	15003	13.343	0.0001	9791	0.0001
Res	500	5.62E+05	1124.4				
Total	524	9.10E+05					

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developments. The investigation has shown that the composition of benthic communities is stable over an approximate 1 m/s range of velocities in high velocity flow environments and that, hence, the effects of tidal energy devices on benthic communities in high velocity environments at far-field scales is likely to be minimal.

Conclusion

At the far-field scale (small water body or environmental impact assessment study area), the relationship between macrobenthic community structure and flow dynamics is likely to depend on local topography, tidal currents and other site specific factors. In this study we have demonstrated a method for high energy epifaunal communities with high levels of natural variability that uses robust replicated analysis to determine the local linkage between flow and community structure that applies to the water body in question. We suggest that studies of this type be incorporated into the ecological assessment of future proposed tidal energy developments. In situ work provides real time information of the system and is highly important to inform and make ecological recommendations for regulators and developers. This would enable environmental mangers to assess the likely impacts of energy removal *per se* on the local communities.

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Author Contributions

Conceived and designed the experiments: LK BE DS GS RK.

Performed the experiments: DS LK.

Analyzed the data: JOC RK DS.

Contributed reagents/materials/analysis tools: BE RK.

Wrote the paper: LK BE RK DS JOC GS.

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Article Chapter V

FLOWBEC: Assessing spatial variation in an epifaunal community in relation to the turbulent wake created by a tidal turbine.

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FLOWBEC: Assessing spatial variation in epifaunal communities in relation to the turbulent wake created by a tidal stream turbine.

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Abstract

The effects of modified flow on epibenthic boulder reef communities adjacent to the SeaGen, the world's first grid-compliant tidal stream turbine, were assessed. The modified wake of the SeaGen was modelled and the outputs were used in conjunction with positional and substrate descriptor variables, to relate variation in epibenthic community structure to the physical environment. An Artificial Neural Network (ANN) and Generalised Linear Model (GLM) were used to make predictions on the distribution of Ecological Status (ES) of epibenthic communities in relation to the turbulent wake of the SeaGen. ES was assigned using the High Energy Hard Substrate (HEHS) Index. ES was largely High throughout the survey area and it was not possible to make predictions on the spatial distribution of ES using an ANN or GLM. Spatial pattern in epifaunal community structure was detected when the study area was partitioned into three treatment areas: area D1; within one rotor diameter (16m) of the centre of SeaGen, area D2; between one and three rotor diameters, and area D3; outside of three rotor diameters. Area D1 was found to be significantly more variable in terms of epifaunal community structure, bare rock distributions and EQR values. However, this variability does not proliferate outside the immediate vicinity of the devices physical footprint.

1 Introduction

Significant changes in energy production methods are required if recent global targets for reducing CO_2 emissions are to be met (IPCC, 2016). The environmental impact of marine renewable energy production is significantly less when compared to carbon or nuclear based energy production methods (Inger et al., 2009). Tidal energy extraction is a relatively novel form of anthropogenic interaction with the coastal environment and its effects on the benthos are still not fully understood (Miller et al., 2013, Sheehan et al., 2013, Broadhurst and Orme, 2014, Kregting et al., 2016, O'Carroll et al., 2017).

Optimal sites for tidal energy extraction are characterized by high tidal velocities and seafloors dominated by rocky outcrops, boulders and coarse sediments (Marine Current Turbines, 2006, 2010; Tidal Energy Limited, 2009; MeyGen, 2011; Sheehan et al., 2013; Broadhurst and Orme, 2014). The SeaGen is the world's first grid-compliant tidal turbine with rated output of 1.2 MW (MacEnri et al., 2013). It is situated in the Strangford Lough Narrows (See Figure 1), a 1 km wide tidal rapid that is dominated by a glacial boulder drop field characterized by a unique biotope under the European Nature Information System (EUNIS) classification scheme (Savidge et al., 2014). Strangford Lough is designated under Natura 2000, it is both a Special Area of Conservation (SAC; UK0016618) under the Habitats Directive (HD: 92/43/EEC) and a Specially Protected Area (UK9020111) under the Birds Directive (BD: 79/409/EEC). The subtidal boulder fields of the Narrows are included as part of the Reefs (1140) qualifying interest for the SAC.

Tidal energy turbines actively remove energy from the environment and the hydrodynamic perturbations created result in the formation of a turbulent wake on the leeward side of the device (Batten and Bahaj, 2006). This turbulent wake may have the potential to negatively affect epibenthic communities through scouring and increased physical disturbance, yet this hypothesis has yet to be definitively assessed.

Variation in epifaunal community structure on boulders is attributed to physical disturbance in the form of the overturning of substrates and scouring by suspended sediment (Palmer and Palmer, 1977, Daly and Mathieson, 1977, Osman, 1977, Sousa, 1979a, Sousa, 1984, McGuinness, 1987b, McGuinness, 1987a, Chapman, 2002a, Chapman, 2002b, Chapman, 2003, Chapman, 2005, Chapman, 2007). Most studies that have assessed community structure on boulders have focussed on intertidal and shallow subtidal habitats. Comparatively little literature exists for epifaunal communities on boulders in the lower infralittoral to the circalittoral (O'Carroll et al., 2017, Kregting et al., 2016).

In high energy environments such as tidal rapids, variability in community structure is more likely to be influenced by physical processes rather than biological interspecific interactions (Sousa, 1984, Dean and Connell, 1987a). Communities that are governed by physical processes tend to exhibit highly variable community structure (Berlow, 1997). The physical processes that govern variability on subtidal rock and boulders have the potential to be amplified in intensity within the wake created by a tidal energy turbine, potentially damaging or removing resident epifauna (O’Carroll et al., 2017, Miller et al., 2013).

Animal-flow interactions are variable and scale dependent (Vogel, 1977, Okamura, 1984, Okamura, 1985, Okamura, 1988, Wildish et al., 1987, Leonard et al., 1988, Patterson, 1991, Hentschel and Herrick, 2005, Hentschel and Larson, 2006), with the highest level of variability existing at the faunal level (Underwood and Petraitis, 1993, Underwood and Chapman, 1996, Wootton, 2001, Terlizzi et al., 2007). Current speed can directly influence the structure of benthic assemblages over a variety of spatial scales, from the feeding-polyp level (Patterson, 1991) to community (Wildish et al., 1987) and habitat levels (Fonseca et al., 1983, Fonseca and Kenworthy, 1987). Patterns in these interactions can be more apparent when observed at coarse ecological resolutions such as ecological functional groups (Warwick and Uncles, 1980, Sebens and Johnson, 1991, Leichter and Witman, 1997, Gili and Coma, 1998).

The epifauna of the Narrows inhabit a stressful environment relative to most other coastal marine habitats. The dynamic physical environment of the Narrows is likely to introduce high amounts of natural variability into the structure of the epifaunal community. High levels of background variability in community structure can make it difficult to detect deleterious change (Underwood, 1994, Underwood, 1991, Underwood and Peterson, 1988). The issue of highly variable community structure masking disturbance effects in conservation exercises has been circumvented through the use of multimetric indices (Forde et al., 2015, O’Carroll et al., 2017, O'Carroll et al., 2016).

Multi-Metric indices such as IQI were developed in response to the Water Framework Directive (WFD; 2000/60/EC). ES is assigned through the assessment of biological, hydromorphological and physicochemical quality elements. An Ecological Quality Ratio (EQR) is derived by comparing putatively disturbed conditions to reference undisturbed conditions (Borja et al., 2000, Borja et al., 2007). An EQR (a decimal value between 0 and 1) is derived by comparing monitoring data from a putatively impacted area to reference conditions. Values close to 1 indicate an ES of “High” and values close to 0 indicate an ES of “Bad”. There are five ES classes (Bad, Poor, Moderate, Good, High), each represents a range of values that exist between class boundary cut-off points along the full EQR value range. The ‘Moderate/ Good’ boundary is considered the most significant threshold under the WFD. Any waterbody classified as ‘Moderate’ or less must be placed under remedial management action plans for impact mitigation, at the expense of the member state (Quintino et al., 2006).

Multimetric indices were initially developed for use in low energy subtidal soft sediments. However, they have been shown to be flexible in their application in different habitat types (Van Hoey et al., 2007, Fitch and Crowe, 2010, Kennedy et al., 2011, Forde et al., 2013, Fitch et al., 2014, Forde et al., 2015), in different geographical regions (Forde et al., 2013), amenable to modification (Forde et al., 2013, Forde et al., 2015) and robust to changes in sampling methodologies (Kennedy et al., 2011). Multimetric indices have been used in monitoring studies that cover a wide array of anthropogenic pressures ranging from sewage and waste water outfalls to various modes of aquaculture (Rhoads et al., 1978, Borja et al., 2000, Van Dalfsen et al., 2000, Borja et al., 2003, Muxika et al., 2005, Bouchet and Sauriau, 2008, Borja et al., 2009, Kennedy et al., 2011, Forde et al., 2015).

The facilitation successional model (Connell and Slatyer, 1977) is central to many multimetric indices and is outlined by Pearson and Rosenberg (1978) for the subtidal sedimentary environment. Pearson and Rosenberg’s (1978) model describes how benthic communities change in a predictable manner along a gradient of decreasing organic loading. As the sediment’s interstitial oxygen levels increase so too do the stress intolerant animals relative to stress tolerant first order opportunists. Within the subtidal reef environment, patches of totally denuded bare rock are analogous to anoxic sediments in that they are the successional start point for reef communities (Chapman, 2002a, Chapman, 2002b, Chapman, 2003, Chapman, 2007). It is from this successional start point that an epifaunal community develops (Chapman, 2007, Chapman, 2003, Chapman, 2002a, O'Carroll et al., 2017).

A WFD compliant multimetric index, the High Energy Hard Substrate (HEHS) index, has been applied to epifaunal dominated reef communities in Strangford Lough and was shown to respond intuitively when the epifaunal community was impacted by natural events at a control station (O'Carroll et al., 2017). The HEHS index was developed to assess the impact of increased physical disturbance on epifaunal dominated reef communities. WFD criteria require multimetric indices to measure 'the level of diversity and abundance of invertebrate taxa' and the 'proportion of disturbance sensitive taxa'. The HEHS index incorporates the Shannon Weiner index (H') (Shannon and Weaver, 1949) and number of species (S) which address 'the level of diversity and abundance of invertebrate taxa' and the Massive/ Crust ratio (Figure 4), which addresses the 'proportion of disturbance sensitive taxa' (See Equation 1). Equation 1 incorporates each metric as a ratio of the observed value to that expected under reference conditions and in doing so meets the requirements of the WFD.

Central to the proposed successional model in Figure 4 is the assumption that on tide swept, stable, reef substrates, massive taxa are 'disturbance sensitive'. While massive taxa are deemed as 'disturbance sensitive', this is a relative term. Both the massive and encrusting taxa within the Narrows are subjected to high ammounts of scouring by suspended sediment twice a day. The spatial extent of the hydroid and sponge dominated community within the Narrows shows that both massive and encrusting taxa are inherently tolerant to these high levels of physical stress. However, the massive taxa exhibit morphological characteristics that are not as robust to physical disturbance events as encrusting taxa.

Massive taxa have a morphological tendency to protrude from the substrate into the water column. They also have a small proportion of their body mass in contact with the substrate. Having such a small point of attachment could result in disturbance events such as brief periods of very high frictional flow (Vogel, 1996), overturning, physical abrasion via mobile gravel cobbles or scouring by suspended sediment (Palmer and Palmer, 1977, Daly and Mathieson, 1977) being catastrophic for the entire animal/ colony if the holdfast is damaged. In the case of encrusting taxa, their morphology reduces the amount of surface area that can be subjected to frictional stress by flow (Vogel, 1996). If an encrusting taxa covers an area greater than that which is affected by a disturbance event, small patches of the colony will remain to recolonize the substrate (Osman, 1977). This pertains to the traditional successional paradigm that a disturbed area will host higher abundances of stress tolerant opportunistic taxa then sensitive taxa (Morris and Wood, 1989, Pearson and Rosenberg, 1978, Sousa, 1979b, Sousa, 1979a, Grall, 1997, Glémarec, 1981).

Integrated environmental monitoring techniques are being employed to advise on the strategic management of the marine environment (Borja et al., 2000, Dowd, 2005, Reiss et al., 2009, Borja et al., 2009, Phillips et al., 2014). For integrative monitoring methods to be effective they must incorporate variables that have high explanatory power for the environment in question (Borja et al., 2003, Phillips et al., 2014). Multivariate community structure, bare rock and HEHS index EQR values have been shown to hold high explanatory power for epifaunal distributions on boulders (Chapman, 2007, Chapman, 2003, Chapman, 2002a, O'Carroll et al., 2017). In this study, we assess the spatial variability of these three metrics to test if the turbulent wake created by the SeaGen is adversely impacting the 'subtidal rock and boulder' qualifying interest of the Strangford Lough SAC.

Study aims:

- To assess the ES of the epifaunal boulder reef communities adjacent to the SeaGen.
- To assess if a predictable relationship exists between the spatial distribution of ES and the turbulent wake created by the SeaGen.
- To assess the variability in established measures of epifaunal community structure and integrity, in a radial manner, around the SeaGen.
- To demonstrate the potential the High Energy Hard Substrate Index has for indicating when significant thresholds in the structural integrity of epifaunal communities have been breached.

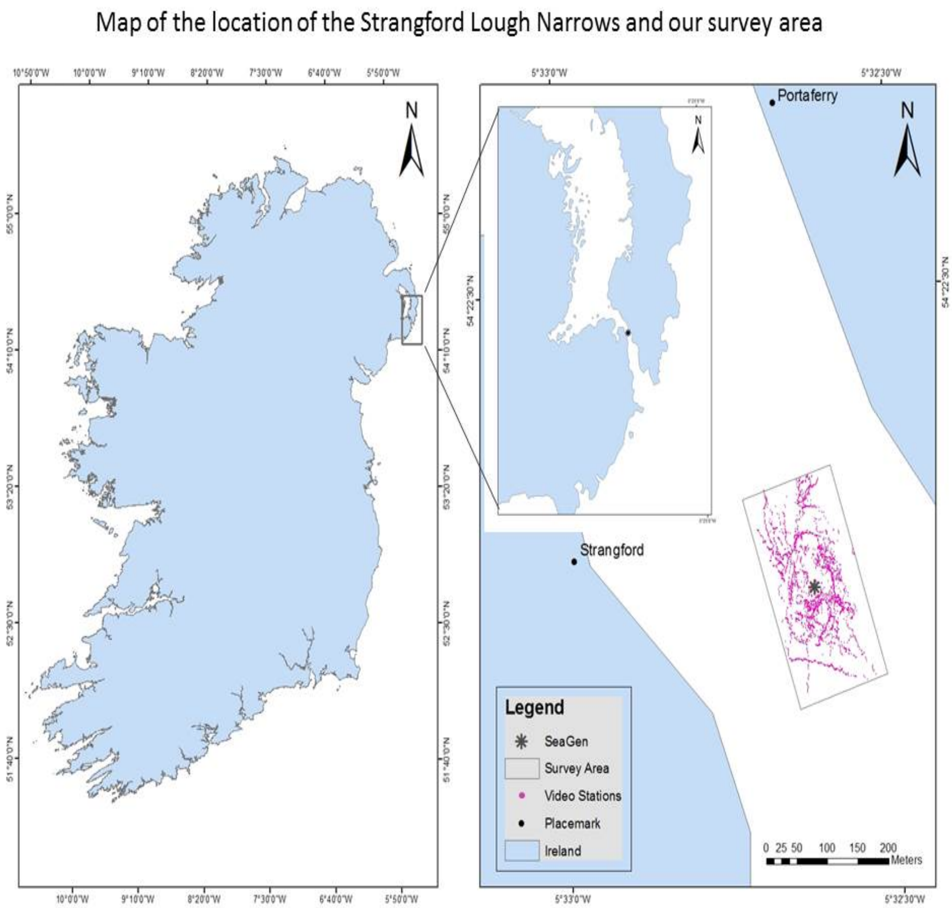

Figure 1: Map of Ireland, the Strangford Lough Narrows, the survey area, position of the SeaGen and the georeferenced drop-down video stations.

2. Materials and Methods

2.1 Field survey

The field survey was carried out over two legs, from the 14th to the 20th of May 2013 and from the 1st to the 2nd of June 2013. The first leg involved a high resolution drop-down video survey of the seafloor around the SeaGen. The second leg consisted of a scuba survey of the seafloor at the foot of the SeaGen using the same drop down video camera frame. The survey was conducted within a 300m X 150m rectilinear polygon (See Figure 1). Video transects were run during slack water on neap tides. A total of 20 drop-down video transects were surveyed. Three diver video surveys were carried out to ensure good complementary coverage of the seafloor at the foot of the device. All observations were made in depths between 20m and 30m.

A high frame rate (50 frames per second) high definition camera system was used to obtain video footage of the seafloor around the device. A Sonardyne Scout USBL (Ultra Short Baseline) system was used to record the position of drop down video system in conjunction with a differential geographic positioning system (DGPS). This provided georeferenced time stamps for each second of video footage subsequently anaysed. This system provided sub one meter location accuracy. The video footage was checked for clarity of focus each time the camera came to rest, to verify that the footage was useable for analysis.

2.2 Video analysis

Video analysis was carried out on stills and footage using VLC media player. The computer screen was divided into one hundred cells using a grid printed on acetate which was placed on the front of the monitor. The area of the seafloor captured by the camera system at any point was on average 50cm^2, determined by scale lasers mounted on the camera frame. A grid with 100 cells was printed on an acetate sheet. Each cell covers approximately 0.5cm^2 of substrate. This allowed for the estimation of percentage faunal cover and the estimation of substrate type distribution (e.g. percentage of gravel, cobbles, boulders and bare rock). Over 3,500 video stills were analysed for this study.

2.3 Substrate type classification

Substrate type classification was carried out *sensu* Eleftheriou and McIntyre (2005). The pebble fraction lies between 8 and 32mm, cobbles between 32 and 256mm with boulders being > 256mm. Bedrock was relatively common and was recorded as such when the camera system was visibly recording over an extensive area of rock substrate, that occasionally had other substrate types present on its surface. The percentage of bare rock was calculated when present within video stills. This is an important measurement within reef environments as it is the successional start point for epibenthic reef communities and an indicator of disturbance (Underwood and Chapman, 1996, Chapman, 2003, O’Carroll et al., 2017).

2.4 Hydrodynamic model

A CFD simulation of a tidal turbine similar to the SeaGen device was developed at the University of Edinburgh, with a solid structure and rotors modelled with the actuator line technique (Creech, 2014, Creech, 2016). The device was modelled in an idealised tidal channel domain 1 km long, 200 m wide, and 30 m deep.

No calibration data was available for the CFD simulation used in this study, but the empty idealised channel model (without the presence of the SeaGen) used velocity and Reynolds Stress profiles from Nezu and Nakagawa (1993), and Stacey et al (1999) for turbulent channels. Turbulence lengthscales were taken from Milne (2011) which were in broad agreement with Nezu and Nakagawa (1993). The resulting velocity profile agreed well with the power law profile for turbulent channels, with an exponent close to the values given for Strangford Narrows (Bearhop et al, 2014).

The CFD simulation used in this study included a dual rotor, contra-rotating tidal turbine structure (Creech, 2014, Creech, 2016) and generated turbulence intensity at the inflow boundary using the Synthetic Eddy Method (Jarrin, 2006), and modelled using Large Eddy Simulation (LES) (Deardroff, 1970) which captured detailed transient structures in the turbulence. Turbulence intensity is a measurement of the temporal fluctuation of flow expressed as a percentage, where smooth, laminar flow has 0% turbulence, and highly turbulent flow could have up to 20% turbulence intensity (http://www.cfd-online.com/Wiki/Turbulence_intensity). The hydrodynamic data was accurately simulated for each georeferenced drop down video station. The coordinates of each drop-down video station relative to the SeaGen (X= 0m, Y= 0m) was calculated in ArcGIS, and then inserted into the CFD model and used as specific simulation nodes. This allowed for the numerical study of the turbulent wake of the turbine in the exact locations where biological and substrate data were derived from video still analysis. The CFD model outputs used in the ecological analyses were time-averaged data for average current velocity (Vel: See Figure 2) and turbulence intensity (Turb: See Figure 3) at 1m above the sea floor at each georeferenced dropdown video station.

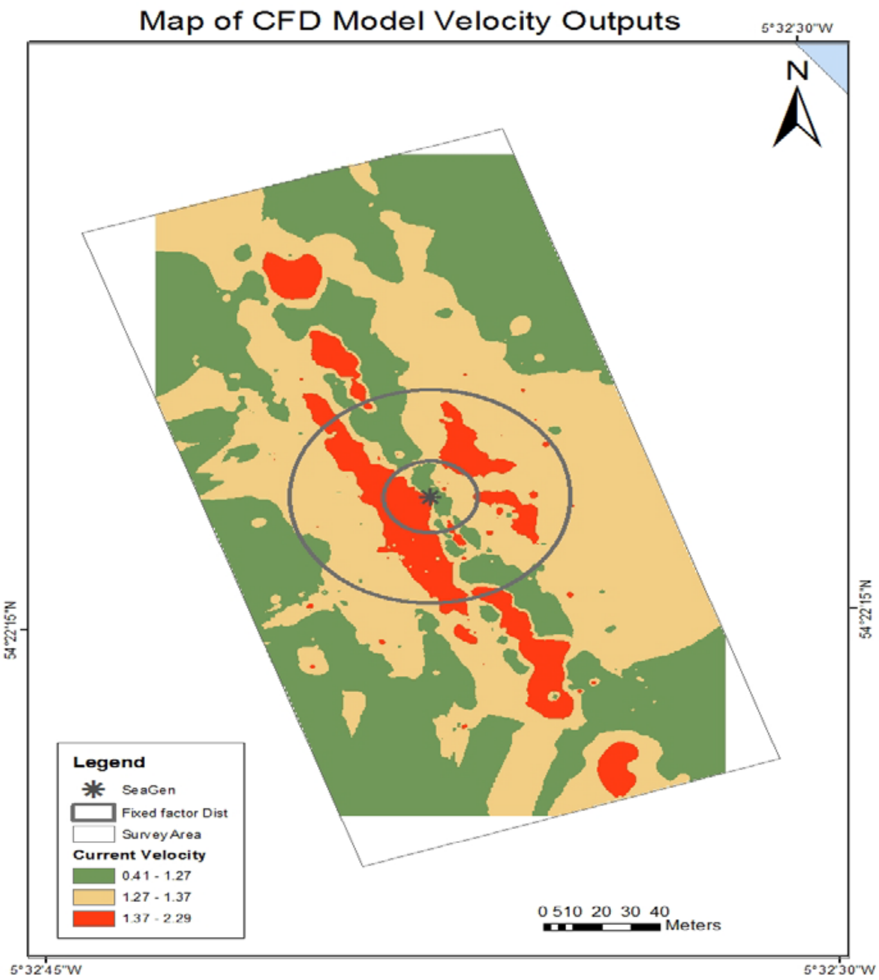

Figure 2: Map of average current velocities in the Narrows. To the North of the SeaGen (centre) are the average current velocities predicted to occur at 1m above the seafloor during a flood tide. Average current velocities at 1m above the seafloor during an ebb tide are depicted to the South of the SeaGen.

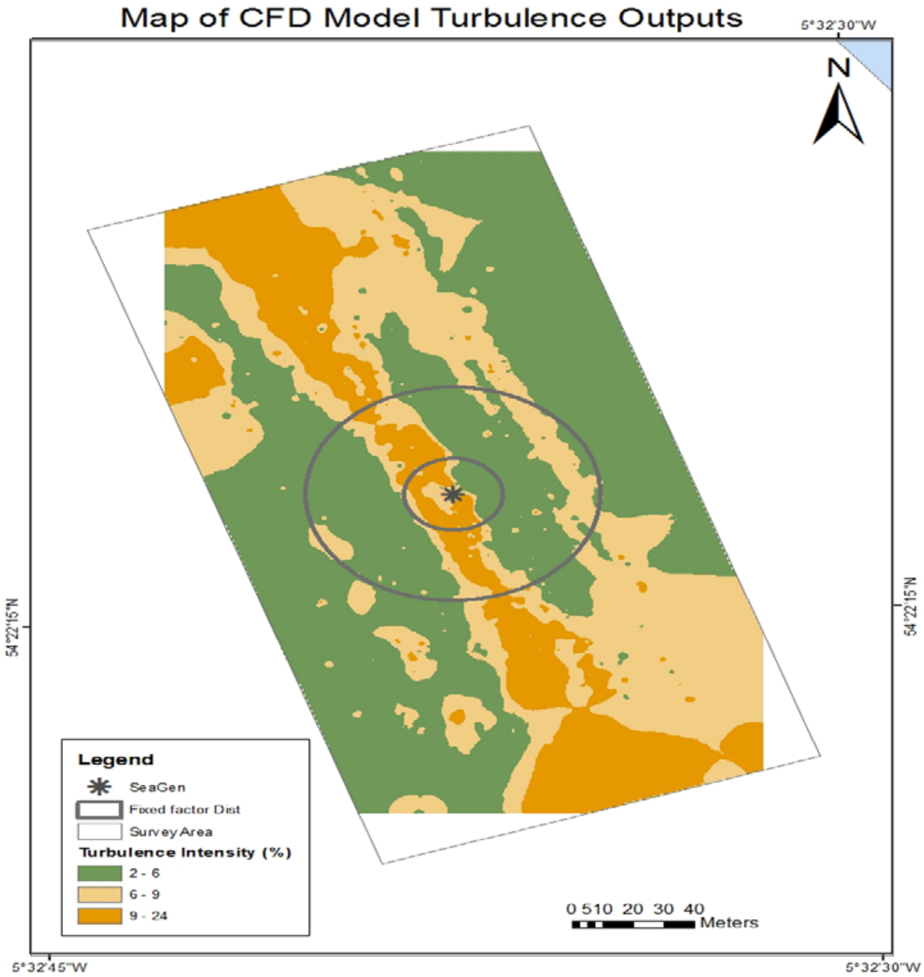

Figure 3: Map of turbulence intensities within the narrows. To the North of the SeaGen (centre) is the predicted turbulence intensities experienced 1m above the seafloor during a flood tide. Turbulence intensities at 1m above the seafloor during an ebb tide are depicted to the South of the SeaGen.

2.5 The High-Energy Hard-Substrate index (HEHS)

A subset of WFD compliant metrics were chosen so that when combined, encompass a high amount of information on how epifaunal communities change within the marine environment. The HEHS index is calculated as shown in Equation 1. Each metric is expressed as a ratio of the maximum observed values from reference stations. Reference values were taken from stations that lay outside of the leeward sides of the SeaGen, areas that would not be influenced by a stream-wise orientated turbulent wake. Bare rock is included as a metric because it is the successional start point for reef communities and its' presence serves as a proxy measure for the extent of a physical disturbance event. An area of 100% bare rock, devoid of any colonising fauna is viewed as totally disturbed and will receive an EQR of 0 (See Figure 4).

Sessile epifauna were placed into two ecological functional groups, massive taxa and encrusting taxa *sensu* Conor et al., (2004). The successional model depicted in Figure 4 asserts that massive species are more sensitive to adverse, physically stressful conditions then encrusting taxa. An area with a high massive to encrusting taxa ratio will receive a higher EQR value than one dominated by encrusting taxa if levels of bare rock, species abundance and diversity are equal in each case.

The HEHS equation can only be applied to animal dominated, stable reef substrates from the lower infralittoral to the circalittoral. This index is not applicable to potentially mobile substrates. When multiple substrate types were present within a video still, the stable substrate(s) was identified and the proportion of the still they occupy was calculated. Percentage faunal cover and bare rock were then calculated as percentages of the subset of the video still. We recommend an acetate sheet divided into 100 cells be custom made to fit the monitor and aspect ratio of the video for ease of percentage cover estimation. All mobile taxa were omitted from the data when applying the HEHS equation as they can actively avoid unfavourable conditions and are not indicative of long term ambient conditions in an area (Terlizzi et al., 2007).

The HEHS index

HEHS=

$$\frac{\left(-\left(\frac{Pi\ BR}{Max\ BR}\right)\times 0.3\right)+\left(\left(\left(\frac{H'}{H'\ max}\right)\times 0.1\right)\right)+\left(\left(\left(\frac{S}{S\ max}\right)\times 0.1\right)\right)+\left(\left(\frac{\left(\left(\frac{\%\ massive}{Max\ obsv.\%\ masv.cover}\right)+1\right)}{\left(\left(\frac{\%\ crust}{Max\ obsv.\%\ crust\ \ cover}\right)+1\right)}\right)\times 0.3\right)}{0.5}$$

(Equation 1)

Where; *Pi BR* = Decimal proportion of bare rock
H’ = Shannon Wiener diversity index

S = Number of species
% massive = Percentage coverage of massive taxa
% crust = Percentage coverage of encrusting taxa
max = The maximum value for the metric from reference conditions

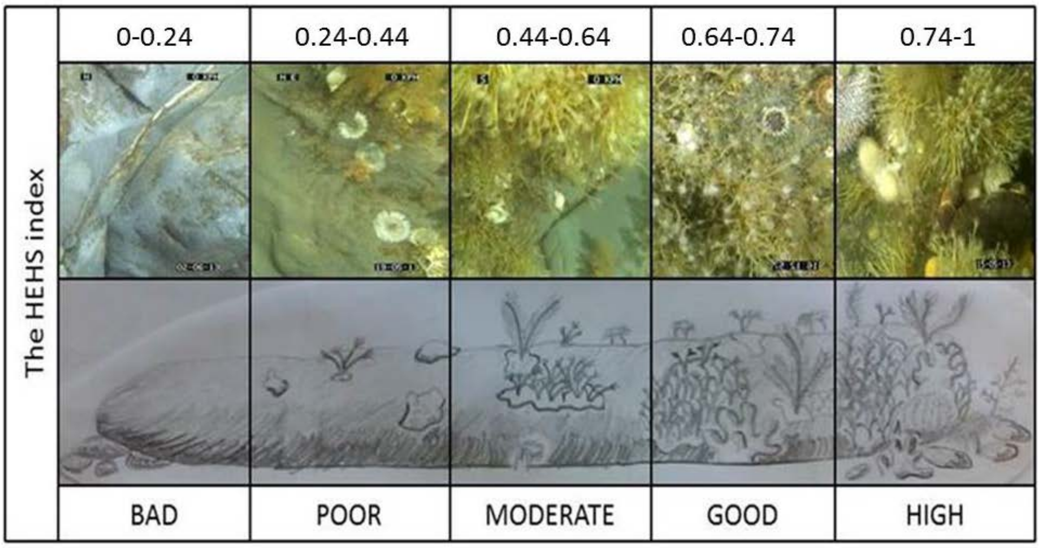

Figure 4: This figure depicts the successional paradigm on which the HEHS is based. The successional change gradient is divided into five parts by class boundary cut-off points at four intervals between 0 and 1. Bad= 0-0.24, Poor=0.24-0.44, Moderate=0.44-0.64, Good=0.64-0.74, High=0.74-1.

2.6 Statistical analysis

All mobile fauna were removed from the abundance matrices prior to statistical analysis (Fraschetti et al., 2005; Terlizzi et al., 2007). Video stills that contained sediment or unstable substrate were removed from the dataset. Only data from stable boulder tops and bedrock were included in the analyses. Substrate mobility and changes in orientation may lead to variability in community structure (Chapman, 2002a, Chapman, 2002b) that is not related to the proximity of the turbine. The percentage coverage of each taxa was square root transformed prior to analysis. See Tables 1 and 2 for a summary of variables, statistical analyses and the individual aims of each analysis. Environmental data was assigned to stations using the spatial join tool in ArcGIS. This tool joins adjacent environmental data to georeferenced stations by appending attributes to the stations attribute table.

Table 1: Summary of the environmental predictor variables and response variables that were used in statistical analyses.

Environmental predictor variables	**Data type**
Distance from the SeaGen measured in rotor diameters (Dist)	Ordinal (3 levels)
Time averaged current velocity (Vel)	Ordinal (3 classes)
Time averaged turbulence intensity (Turb)	Ordinal (3 classes)
Boulder tops	Binary
Bedrock	Binary
Stream-wise positioning of a station relative to the SeaGen (Stream)	Binary

Response variables	**Data type**
ES of a station along the critical Moderate/Good boundary (Binary ES)	Binary
ES of a station (Ordinal ES)	Ordinal (5 classes)
Zero-adjusted Ranked Bray-Curtis (BC; 1957) similarities of square root transformed epifaunal coverage data (Multivariate Fauna)	Multivariate ordinal
Percentage bare rock (Bare Rock)	Continuous (0-100%)
HEHS Index EQR values (EQR)	Continuous (0-1)

Table 2: Summary of statistical analyses and the associated aim of each analysis. Note: Analyses used in EUNIS biotope classification are excluded.

Summary of statistical analyses and individual aims

Analysis	**Aim**
GLM	To assess the relationship between the binary ES response variable and the environmental predictor variables
multinom	To assess the relationship between the 5 ES classes and the environmental variables
BVSTEP	To test if the distribution of a species/ subset of species can explain pattern in multivariate community structure.
PERMANOVA	To test the effects of the levels of the fixed factor Dist on Bare Rock, EQR values and Multivariate Fauna
SIMPER	To outline compositional differences between levels when a significant effect of Dist was detected
PERMDISP	To test the homogeneity of multivariate faunal, bare rock and EQR dispersions within each level of the fixed factor Dist

2.6.1 Assessing the relationship between epifaunal community structure and the turbulent wake of the SeaGen: Generalised Linear Modelling (GLM) Artificial Neural Networks (ANNs)

A GLM using the binomial distribution, logistic link function and a mixture of forwards and backwards selection was used to relate binary ES responses (0= Moderate or less, 1= Good or High) to the environmental predictor variables. The diameter of one of the SeagGens' rotors is 16m. Distance (m) from the turbine (Dist) was used as a nominal variable and treatment areas were divided using rotor diameters as distance units (3 treatment areas; D1 within one rotor diameter, D2= between 1 and 3 rotor diameters. D3= more 3 rotor diameters away from the device). Hydrodynamic variables (Vel and Turb) were used as nominal variables and were divided into 3 classes using natural breaks, or Jenks, in ArcGIS. This method selects 'natural' groups of values based on the distribution of Vel and Turb magnitudes along an X-axis. The class boundary cut off points were determined by selecting asymptotes in a trimodal distribution curve fitted to the distributions of the Vel and Turb values observed. The presence or absence of boulder tops and bedrock were included as binary substrate type distribution variables. A binary variable (Stream) was used to describe whether a station was upstream or downstream of the SeaGen. The variable 'Stream' was included to help detect if the SeaGen was significantly affecting communities upstream and downstream in an asymmetrical manner.

An ANN was created using multinomial regression analysis which made predictions on the distribution of the five ES classes using the environmental predictor variables. ES classes were converted into an ordinal response variable with 5 levels. A single hidden-layer, feed-forward neural network was constructed using the multinom package (Venables and Ripley, 2002) in the R statistical software environment v3.3.1 (R Development Core Team, 2014). Random unstratified selection of stations was used to create the training datasets.

2.6.2 EUNIS biotope classification and assessment of variance in epifaunal community structure

Multivariate statistical analyses were carried out using PRIMER V6 (Clarke and Gorley, 2006, Clarke et al., 2006). A zero-adjusted BC similarity matrix (Bray and Curtis, 1957, Clarke et al., 2006) was compiled for the transformed faunal data. Habitat classification at each station was performed in accordance with the 04-05 JNCC habitat classification scheme (Conor et al., 2004). Hierarchical clustering and Similarity Profile (SIMPROF) analyses (Clarke et al., 2008) were carried out on the zero-adjusted transformed resemblance matrix. SIMPROF analysis is a test of the null hypothesis of no significant group structure within the faunal matrix. It produces groups of samples that are significantly different to other groups within the dataset. SIMPER analysis (Clarke, 1993) was performed to determine the characterizing species for each group identified by SIMPROF. These data were compared to the biotope descriptions of Conor et al. (2004) to determine which biotope the groups corresponded to.

BVSTEP analysis (Clarke and Ainsworth, 1993) was used to test if the distribution of a single species or a subset of species could explain pattern in multivariate community structure. This procedure was used to test for high rank correlations between the square root transformed faunal abundance data matrix and its own BC resemblance matrix. Weighted Spearman was used as the correlation coefficient.

2.6.3 Assessing the effect of distance from the SeaGen on epifaunal community structure

A one way permutational analysis of variance (PERMANOVA; Anderson, 2008) was carried out on the Multivariate faunal data using distance from the turbine (Dist) as a fixed factor. The same analysis was repeated using Euclidean distance based matrices for both Bare Rock and EQR values at each sample station. Tests for main effects and pairwise differences were performed in each case. SIMPER analysis was used to outline compositional differences between treatment levels when a significant effect of Dist was detected.

PERMDISP (Clarke and Gorley, 2006) was used to test for variability in the distributions of multivariate faunal, bare rock and EQR value data within each level of the fixed factor Dist. PERMDISP is a multivariate equivalent to Lavene's test (Levene, 1960), as it is a test of homogeneity of multivariate/ univariate dispersions among groups, based on distance or (dis)similarity measures. Increases or decreases in homogeneity/ variability can be interpreted as potential indicators of stress in response to environmental impacts. PERMDISP is also a logical compliment to PERMANOVA to determine whether dispersions are contributing to significant differences detected by PERMANOVA.

3 Results

Table 3: Summary of the results of statistical analyses. Note: The analyses used for EUNIS biotope classification are excluded.

Summary of the results of statistical analyses

Analysis	Result
GLM	Could not predict binary ES distributions using the environmental variables
ANN	Could not predict ordinal ES distributions using the environmental variables
BVSTEP	The distribution of a subset of dominant fauna explains 95% (Rho= 0.955, p=<0.001) of variability in community structure
PERMANOVA	The fixed factor Dist had a significant effect on Multivariate fauna, Bare rock, and EQR distributions
SIMPER	Treatment D1 had the highest amount of bare rock, the lowest EQR and the lowest relative abundances of the dominant, erect species
PERMDISP	Treatment D1 had the highest dispersions of Bare Rock, EQR and Multivariate Fauna values around group means than the other treatments D2 and D3

3.1 Hydrodynamic model outputs

According to the CFD simulation outputs, the average current velocity 1m above the seafloor in the survey area is 1.3ms^{-1}, minimum current velocity is 0.4ms^{-1} and the maximum current velocity is 2.3ms^{-1}. 99.6% of stations had average current velocities of between 1ms^{-1} and 1.7 ms^{-1}. There was a large range of turbulence intensities within the survey area. The maximum turbulence intensity was 24%, average turbulence intensity was 8.5%, and the minimum turbulence intensity was 2%.

3.2 Assessing the relationship between ES and the turbulent wake of the SeaGen

The HEHS Index classified majority of the survey area as High (88% of stations) or Good (8% stations). The remaining 4% of stations were classified as Moderate or less and were largely clustered at the foot of the SeGen (See figure 5). The GLM or ANN could not make correct predictions on the spatial distribution of ES classification distributions to environmental variables (See Table 4A, 4B and 4C). In no instance did the inclusion of predictor variables improve on the Akaike Information Criterion (AIC; Akaike, 1973) of the intercept in the GLM (Table 4A). The multinomial model did not find any predictor variables to significantly improve the intercept AIC, or successfully reclassify any of the ES classifications. In both cases, all the predictor variables were included in the optimal models. Both models were based on lowest AIC (See Table 4C).

Table 4: Output of ANN and GLM analyses. (A) Results of the GLM using binary ES classifications. (B) Results of the multinomial model and (C) the goodness of fit table for the multinomial model in which all responses were classified as High (1).

(A) Generalised Linear Model outputs

	Df	Deviance	AIC
Intercept		290.23	292.23
Turb	2	286.38	292.38
Boulder	1	290.12	294.12
Bedrock	1	290.21	294.21
Stream	1	290.22	294.22
Sist	2	289.32	295.32
Vel	2	289.88	295.88

(B) Multinomial model outputs

	Df	AIC	Pr(Chi)
Intercept		771	
Boulder	4	763	0.99
Bedrock	4	770	0.13
Stream	4	764	0.77
Dist	8	758	0.93
Vel	8	756	0.99
Turb	8	764	0.3

(C) Goodness of fit table for multinomial model

	Fitted				
Given	*1*	*2*	*3*	*4*	*5*
1	621	0	0	0	0
2	48	0	0	0	0
3	13	0	0	0	0
4	16	0	0	0	0
5	8	0	0	0	0

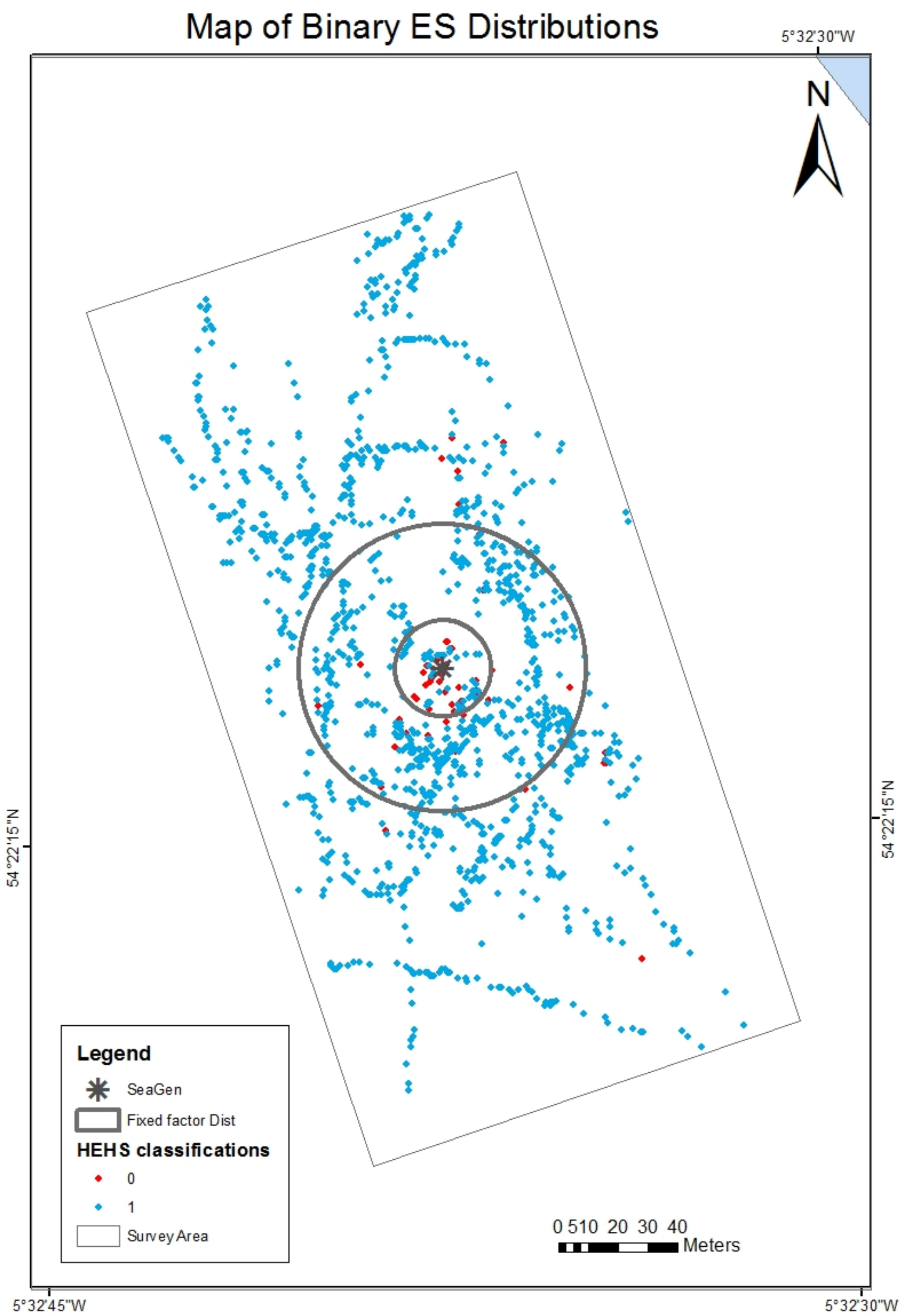

Figure 5: Map of the distribution of ES classifications along the Moderate/Good boundary at stations that exist on boulder tops and bedrock only (Moderate or less= 0, Good or better= 1). The treatment areas of the fixed factor D are also shown.

3.3 Epifaunal community structure and how it is affected by distance from the SeaGen

Epifaunal community structure was highly variable within the survey area. SIMPROF analysis on square root transformed multivariate faunal data created 120 significantly different groups within a 350m x 150m rectilinear polygon. The lowest level of BC similarity within these groups was 38%. SIMPROF groups could only be ascribed to the level-4 biotope complex CR.HCR.Fat; very tide-swept faunal communities *sensu* Conor et al., (2004).

The hydroids *Tubularia indivisa* (Linnaeus, 1758) and *Sertularia argentea* (Linnaeus, 1758), were the most abundant fauna in the survey area. The sponges *Halichondria panicea* (Pallas, 1766), *Amphilectus fucorum* (Esper, 1794) were the most abundant Porifera. The morphological descriptor 'Mixed faunal turf' was also abundant throughout the survey area. The most abundant anthozoans were species of the genus *Sagartia* (Gosse, 1885) and the colonial species *Corynactis viridis* (Allman, 1846). Tunicates such as *Dendrodoa grossularia* (Van Beneden, 1846) and *Ciona intestinalis* (Linenaeus, 1767) were also common. The abundances of the hydrozoan dominants, *Sertularia argentea* and *Tubularia indivisa,* were lower within the treatment area D1 compared to areas D2 and D3 (See Appendix Table A1c).

BVSTEP analysis showed that a subset of three species and a morphological type description of encrusting taxa produced a multivariate distribution that was very similar to the multivariate community structure of the entire dataset (*Tubularia indivisa, Sertularia argentea, Halichondria panicea,* mixed faunal turf*:* Rho=0.955, p=<0.001). The selected fauna are almost ubiquitous across all the stations in the survey area. The distribution of these taxa may be useful as an indicator of overall community structure. *Tubularia indivisa* and *Sertularia argentea* are erect hydrozoans. *Halichondria panicea* and *mixed faunal turf* are both crustose.

A significant effect of the fixed factor Dist on bare rock distributions, EQR values and multivariate faunal data was detected (Table 5 and Appendix Table A2). SIMPER analysis on bare rock showed that the mean percentage bare rock was higher within area D1 than D2 or D3 (Appendix Table A1a). The average squared distance was higher in D1 indicating a more variable distribution of bare rock compared to D2 and D3. EQR values were lowest in D1 and also the most variable (Appendix Table A1b). The average value (EQR=0.567) per station in treatment area D1 falls below the Moderate/Good class boundary *sensu* the WFD, within the immediate footprint (1 rotor diameter) of the SeaGen. Epifaunal community structure was significantly different in D1 compared to D2 and D3 (Appendix Table A2c), and contained the lowest abundances of hydrozoan dominants (Appendix Table A1c).

PERMDISP analysis showed each treatment area to be significantly different in terms of Euclidean distance dispersions around the group mean based on Bare Rock distributions (Appendix Table 3a). The average of all Euclidean distance dispersions around the group mean decreased with distance from the SeaGen (Table 6A).

D1 was significantly different to the treatment areas D2 and D3 in terms of Euclidean distance dispersions around the group mean based on EQR values. D1 was the most variable around its group mean dispersion value. D2 and D3 were not significantly different from each other (Appendix Table A3b). This variability around group mean decreased with distance from the SeaGen (Table 6B).

PERMDISP analysis showed D1 to have the highest dispersion values around its group mean based on BC Multivariate Fauna distributions. D2 and D3 had similar dispersions around their means and were not significantly different from each other (Table A3c). Variability in epifaunal community structure decreased with distance from the SeaGen (Table 6C).

Table 5: Results of PERMANOVA on (A) Bare rock distributions (B) EQR values and (C) Multivariate faunal community structure (Multivariate fauna). The fixed factor Dist had a significant effect on the response variables in all cases.

(A) Bare rock

Source	**df**	**SS**	**MS**	**Pseudo-F**	**P(perm)**	**Unique perms**
Dist	2	4.571	2.2855	26.321	0.001	999
Res	1452	126.08	8.68E-02			
Total	1454	130.65				

(B) EQR

Source	**df**	**SS**	**MS**	**Pseudo-F**	**P(perm)**	**Unique perms**
Dist	2	4.6632	2.3316	197.63	0.001	997
Res	1452	17.13	1.18E-02			
Total	1454	21.794				

(C) Multivariate fauna

Source	**df**	**SS**	**MS**	**Pseudo-F**	**P(perm)**	**Unique perms**
Dist	2	20078	10039	7.9541	0.001	998
Res	1452	1.77E+06	1262.1			
Total	1454	1.79E+06				

Table 6: The tables outline the number of stations within each level of the factor Dist (Size) the average of all multivariate distances from the group centroid for each factor level (Average) and the Standard Error for each factor level (SE). D1 has the largest variance around its mean in each case.

(A) Bare rock

Group	**Size**	**Average**	**SE**
D1	82	0.33	1.89E-02
D2	512	0.25	7.43E-03
D3	861	0.23	5.05E-03

(B) EQR

Group	**Size**	**Average**	**SE**
D1	82	43.842	1.3975
D2	512	31.108	0.6947
D3	861	32.416	0.47468

(C) Multivariate fauna

Group	**Size**	**Average**	**SE**
D1	82	43.842	1.3975
D2	512	31.108	0.6947
D3	861	32.416	0.47468

4 Discussion

Established metrics of epifaunal community structure and integrity for this habitat were found to be significantly more variable within one rotor diameter of the SeaGen. This 'benthic footprint' (Miller et al, 2013) is restricted to the physical footprint of the SeaGen and does not proliferate into the surrounding habitat. This variability in epifaunal community structure, bare rock and EQR value distributions is not considered to be a significant spatial impact due to its restricted spatial extent.

ES distributions could not be attributed to the simulated hydrodynamic parameters. According to the CFD model, hydrodynamic conditions one metre off the seabed showed very little variation, with 99.6% of stations having average current velocities of between $1ms^{-1}$ and $1.7\ ms^{-1}$. Epibenthic communities in this habitat have been shown to exist as normal, stable state communities in current speeds of up to $2.7m^{-1}$ (Kregting et al., 2016). The epifauna in the greater survey area appear resilient against any influence the SeaGen may be having on the physical environment outside of one rotor diameter from its' bottom structure. A significant change in hydrodynamic conditions or sedimentation rates would be necessary to negatively affect benthic communities within this type of high energy environment (Neill et al., 2009, Shields et al., 2011, Miller et al., 2013). Our results suggest that in tidal rapids that contain single devices, these large scale impacts are unlikely to occur.

Assessing community structure in a radial manner around the SeaGen allowed for the spatial pattern in community structure to be detected. Treatment area D1 was found to have significantly more variable community structure than D2 and D3. All treatment areas had epifaunal communities that were significantly different to one another in terms of the relative abundances of dominant species. This suggests that there is a gradual change from altered communities in D1 to ambient 'normal' communities at D3, with communities at D2 existing in an intermediary state.

The increased levels of bare rock in D1 indicated that ambient physical conditions are being influenced by the SeaGen in a way that is detrimental to resident epifauna, as bare rock is an indication of recent disturbance (Chapman, 2002a, Chapman, 2002b, Chapman, 2003, Chapman, 2007). D1 is the only treatment area that has an average ES of Moderate. The “Moderate/Good” boundary is deemed the critical boundary under the WFD, with remedial management action required to restore areas classified as Moderate or worse, to Good or better. According to the HEHS index the SeaGen is having a negative influence on epifaunal communities immediately between and adjacent to the device legs. However, this effect dissipates quickly with increased distance from the device. The HEHS index did not find epifaunal communities within D2 to be affected relative to reference conditions. This could indicate that the intermediary state community in D2 detected by multivariate faunal analyses exists in the realm of natural variability for the habitat.

The three dominant taxa *Tubularia indivisa, Sertularia argentea, Halichondria panicea* and the morphological type description ‘mixed faunal turf’ equally represent the two ecological functional group in the ‘Massive: Crust’ ratio used in the HEHS index. The reduction in massive taxa cover and increase in the percentage of bare rock in D1 pertains to the predictions made in the successional model outlined in Figure 4. Our results suggest that it is likely to be possible to detect spatial effects of tidal energy extraction by assessing dominant faunal types only. This could significantly save on associated monetary costs of monitoring efforts by reducing taxonomic effort.

Using a multimetric index that incorporates species’ tolerance to stress in the sedimentary environment is an effective way of detecting higher order effects that multivariate faunal data and univariate diversity indices may not detect (Borja et al., 2003, Reiss and Kröncke, 2005, Kröncke and Reiss, 2010, Forde et al., 2015). In this study, we could detect significant change in multivariate community structure and bare rock distributions, and produce a quantitative, easily interpretable measurement of this change by using the HEHS index. Another benefit of incorporating a multimetric index in an applied ecological survey such as this, is that EQR values put a quantitative measure on variability in community structure that is otherwise difficult to define in terms of disturbance effects (Borja et al., 2007, Muxika et al., 2007).

This study, and O'Carroll et al. (2017), are the only two fully quantitative assessments of the interaction between an operational tidal stream turbine and the adjacent benthos. The interactions between epifauna and tidal energy extraction reported here, and in O'Carroll et al. (2017), could be site and device specific. There is, however, merit to the approaches adopted in these Strangford Lough case studies. The identified metrics for epifaunal community integrity in this habitat provide an ecologist with the tools to assess change in community structure and functioning in an intuitive manner in relation to increased physical disturbance.

The binary HEHS index (O'Carroll et al., 2017) outputs indicate if a significant threshold of community structural integrity and functioning has been breached. This could also be described as a 'traffic light' method for decision making in relation to established thresholds (Wilding et al., 2017).

Macroscale ecosystem services are ultimately dependant on the fine scale distributions of benthic fauna, as their individual ecological roles culminate to produce significant amounts of secondary production and biogeochemical cycling (Galparsoro et al., 2014). This is reflected in the adoption of the ecosystem based approach to marine conservation in HD, BD, WFD and the Marine Strategy Framework Directive (MSFD; Council Directive 2008/56/EC). These Directives are some of the most progressive legislation on ecosystem-based approaches to marine conservation in the world (Borja et al., 2010). These Directives, in particular the WFD, HD and BD, are implemented through a network of small scale studies, the output of which combines to provide large scale information on the status of the marine environment. The integration of benthic conservation methodologies across the Directives, and the resulting standardisation of outputs, is a goal of the MSFD. Multimetric indices can be used to obtain this goal. The HEHS index is the first attempt at giving subtidal reefs equal representation in terms of benthic monitoring standards and rigour. Its application in tidal rapids outside of Strangford Lough will initially require intercallibration studies (ie. Muxika et al., 2007, Van Hoey et al., 2007).

A quantitative measurement of benthic habitat status at tidal energy extraction sites could be developed at regional scales over time. This will give ecologists the information that is needed to assess the effects of tidal energy extraction on benthos dependant, macroscale ecosystem service provision. Currently, there is not enough data regarding the fine scale interactions between tidal energy extraction and the benthos to make inferences on the potential knock effects for ecosystem service provision in tidal rapids.

The epifaunal community in the survey area was dominated by a small subset of taxa. The relative abundances of these taxa were highly variable and exhibited random local structure in the greater survey area. Our results are in agreement with those of experimental studies that have found the interaction of epifaunal filter feeders and the physical environment to be highly variable (Okamura, 1984, Okamura, 1985, Okamura, 1988). Other field investigations have also found colonisation patterns on boulders in the intertidal and shallow subtidal to be difficult to predict (Chapman, 2002a, Chapman, 2007). None of the aforementioned studies on the colonisation of boulders report on tide-swept subtidal boulder reef environments. Within a dynamic physical environment such as the Narrows, physical processes are likely to override interspecific interactions in structuring the epifaunal community (Berlow, 1997). These physical processes are likely to be driven by abrasive forces that are a function of hydrodynamic conditions (Osman, 1977, Sousa, 1979a, Chapman, 2002a). We found the epifaunal community of the survey area to be a homogenously noisy entity, and our results portray the intrinsic stochastic processes that govern community structure on subtidal boulders.

The interaction between the bottom quadrat of the SeaGen and flow could not be included in the CFD model due to computational difficulties. It is possible that the spatial effect on response variables in D1 is a result of the bottom structure significantly altering hydrodynamic conditions in a manner that pushes them outside of the realm of natural, non-destructuve variation (Kregting et al, 2016).

Given that installation occurred in 2008 and substrates in D1 were still not fully colonised in 2012, it is unlikely that there is a lasting construction effect. To definitively assess the recovery rate of this community, a control area of boulder reef that is spatially distant from the SeaGen would need to be totally denuded and the subsequent colonisation rates monitored. It is most likely that substrate colonisation within the footprint of the SeaGen is being held at an early successional stage as a result of increased physical disturbance.

When compared to certain methods of aquaculture, such as finfish cage culture and high density longline mussel culture, the SeaGen has minimal benthic impacts. Methods of Aquaculture such as finfish cage, net pen culture, and long-line mussel cultures can have an impact on benthic communities directly beneath and downstream of the culture sites (Karakassis et al., 2000, Buschmann et al., 2006, Kalantzi and Karakassis, 2006, Giles, 2008, Borja et al., 2009). This study shows that tidal energy extraction has a negligible influence on the benthic environment. Others have found the benthic effects of marine renewable energy devices to be beneficial to ambient communities (Reubens et al., 2013, Bergström et al., 2013, Langhamer and Wilhelmsson, 2009). Given the knowledge and 'acceptability' of the impacts of aquaculture, there is potential for the significant growth in the production of marine renewable energy at minimal expense to the immediate environment.

The installation of multiple devices in a small area could affect benthic communities in a significant manner. Issues such as the interaction of multiple turbulent wakes and intermediate communities around the devices in a small area will need to be investigated. We recommend that hydrodynamic models developed for use in such studies incorporate bathymetric information as a measure of bottom roughness and as much of the device structure as possible. A spatial resolution of 1m to 2m grid cells would be an adequate scale of hydrodynamic model output for a study of this kind. More research is needed to assess the cumulative benthic effects of multiple operational tidal energy turbines.

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Supplementary material

Table A1: This table summarises nine SIMPER tables of results. SIMPER analysis was used to outline the compositional differences between treatment levels of the factor Dist based on; (a) Bare rock distributions, (b) EQR values and (c) Multivariate faunal community structure. Percentage contribution and cumulative contribution were omitted from tables (a) and (b) as both equalled 100% in each case.

(a) Bare rock

D1			
Average squared distance = 0.14			
Species	Av.Value	Av.Sq.Dist	Sq.Dist/SD
%BR	0.476	0.141	0.54
D2			
Average squared distance = 0.09			
Species	Av.Value	Av.Sq.Dist	Sq.Dist/SD
%BR	0.285	9.44E-02	0.47
D3			
Average squared distance = 0.08			
Species	Av.Value	Av.Sq.Dist	Sq.Dist/SD
%BR	0.294	8.36E-02	0.47

(b) EQR

D1			
Average squared distance = 0.08			
Species	Av.Value	Av.Sq.Dist	Sq.Dist/SD
EQR	0.567	8.39E-02	0.5
D2			
Average squared distance = 0.05			
Species	Av.Value	Av.Sq.Dist	Sq.Dist/SD
EQR	0.694	4.73E-02	0.37
D3			
Average squared distance = 0.04			
Species	Av.Value	Av.Sq.Dist	Sq.Dist/SD
EQR	0.713	3.75E-02	0.33

(c) Multivariate fauna

D1					
Average similarity: 25.25					
Species	**Av.Abund**	**Av.Sim**	**Sim/SD**	**Contrib%**	**Cum.%**
Sertularia argentea	3.88	13.45	0.65	53.27	53.27
Tubularia indivisa	2.79	7.59	0.51	30.07	83.34
Mixed faunal turf	1.25	3	0.33	11.9	95.25
Electra pilosa	0.54	0.75	0.19	2.99	98.23

Halichondria bowerbanki	0.4	0.15	0.1	0.59	98.82
Halichondria panicea	0.45	0.12	0.08	0.48	99.3

D2

Average similarity: 51.98

Species	**Av.Abund**	**Av.Sim**	**Sim/SD**	**Contrib%**	**Cum.%**
Tubularia indivisa	5.52	26.89	1.43	51.73	51.73
Sertularia argentea	4.93	21.6	1.21	41.54	93.27
Mixed faunal turf	0.92	1.72	0.31	3.3	96.57
Halichondria panicea	0.97	1.27	0.29	2.43	99.01

D3

Average similarity: 50.78

Species	**Av.Abund**	**Av.Sim**	**Sim/SD**	**Contrib%**	**Cum.%**
Tubularia indivisa	5.55	25.77	1.39	50.75	50.75
Sertularia argentea	4.89	20.03	1.11	39.44	90.19
Halichondria panicea	1.27	2.18	0.39	4.3	94.49
Mixed faunal turf	1.06	1.78	0.33	3.5	97.98
Alcyonium digitatum	0.43	0.35	0.18	0.69	98.67
Amphilectus fucorum	0.28	0.2	0.16	0.39	99.06

Table A2: Results of PERMANOVA Pair-wise comparisons between the three treatment levels of the fixed factor Dist on (a) bare rock distributions and (b) EQR values and (c) Multivariate faunal community structure. The results in (a) show that all treatment levels are significantly different in terms of their bare rock distributions. The results in table (b) show that the treatment level D1 is significantly different to all other factor levels in terms of EQR values. The results in table (c) show that all the treatment levels were significantly different in terms of their epifaunal community structure. t= The test statistic, P(perm)= P values obtained through permutations, Unique perms= amount of permutations, P(MC)= P value obtained by Monte-Carlo test.

(a) Bare rock

Groups	**t**	**P(perm)**	**Unique perms**
D2, D1	5.4524	0.001	998
D2, D3	2.4042	0.01	996
D1, D3	7.3387	0.001	999

(b) EQR

Groups	**t**	**P(perm)**	**Unique perms**
D2, D1	14.677	0.001	997
D2, D3	0.63755	0.531	999
D1, D3	18.576	0.001	995

(c) Multivariate fauna

Groups	**t**	**P(perm)**	**Unique perms**
D2, D1	3.2499	0.001	999
D2, D3	1.9222	0.012	999

D1, D3	3.5571	0.001	999

Table A3: Results of PERMDISP analysis. (a) All treatment levels were found to be significantly different to each other based on distributions of bare rock around the group mean value. (b) D1 was found to be significantly different to D2 and D3 based on dispersions of EQR values around the group mean. Table (c) shows that D1 is significantly different in terms of its multivariate dispersions of BC similarities around the group mean value.

(a) Bare rock

F: 15.359	**df1**: 2	**df2**: 1452
Groups	**t**	**P(perm)**
(D2,D1)	3.8221	2.00E-03
(D2,D3)	2.3622	3.90E-02
(D1,D3)	5.5919	1.00E-03

(b) EQR

F: 270.88	**df1**: 2	**df2**: 1452
Groups	**t**	**P(perm)**
(D2,D1)	18.673	1.00E-03
(D2,D3)	2.6274	6.50E-02
(D1,D3)	23.557	1.00E-03

(c) Multivariate fauna

F: 27.352	**df1**: 2	**df2**: 1452
Groups	**t**	**P(perm)**
(D2,D1)	6.9804	1.00E-03
(D2,D3)	1.6034	0.162
(D1,D3)	7.1516	1.00E-03

Article Chapter VI

Using Artificial Neural Networks to predict the spatial distribution of EUNIS biotopes in a large embayment

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Using artificial neural networks to predict the spatial distribution of EUNIS biotopes in a large embayment

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Abstract

In this study, feed-forward artificial neural networks (ANNs) were used to make predictions on the spatial distributions of EUNIS level four subtidal sediment biotopes in Galway Bay, Ireland. A cost effective modelling approach was developed with the aim of making it applicable in offshore, MSFD benthic mapping exercises. The INFOMAR (Integrated Mapping of Irelands Marine Resource) initiative has acoustically mapped and classified a significant proportion of Irelands Exclusive Economic Zone (EEZ). As much publically available data as possible was incorporated in our model which included depth and INFOMAR acoustic seabed data, transitional water boundaries and grab survey data. Targeted grab surveys were carried out by the authors to fill any spatial gaps in the data. A hydrodynamic model was developed in-house and is a high resolution 3D numerical model of Galway Bay. The Ecological Status (ES) of the subtidal sediments of Galway Bay was also assessed.

The ES of Galway Bay was mostly 'High' with some stations being classed as 'Good'. To make predictions on the distribution of EUNIS biotopes optimal models were determined using multinomial modelling techniques prior to making predictions using ANNs in R. Optimal models used a combination of salinity, proximity to reef, depth and a sediment descriptor as predictor variables.

ANNs that used observed sediment classes as predictor variables could predict the distribution of biotopes 67% of the time, compared to 63% for ANNs using acoustic sediment classes. Acoustic sediment ANN predictions were affected by local sediment heterogeneity, and the lack of a mixed sediment class. Within Galway Bay, INFOMAR data can be used as a surrogate for traditional sediment

predictor variables and only result in a small reduction in total correct model predictions.

1. Introduction

The extent and trajectory of anthropogenic impacts on the marine environment is reflected in the implementation of progressive, trans-boundary conservation frameworks, such as the Habitats Directive (HD: Council Directive 92/43/EEC), Birds Directive (BD; Council Directive 79 / 409 / EEC), Water Framework Directive (WFD: Council Directive, 2000/60/EC) and the Marine Strategy Framework Directive (MSFD; Council Directive 2008/56/EC).

The Marine Strategy Framework Directive (MSFD) establishes a framework within which EU Member States are required to take the necessary measures to achieve or maintain Good Environmental Status (GES) in the marine environment by 2020. The aim of the MSFD is to protect European marine waters through ecosystem-based management of human activities while enabling the sustainable use of the marine environment for present and future generations. Ireland has the largest seascape to land ratio in Europe and must implement measures to sustainably manage 488,762 km2 of ocean (Barry et al., 2013).

Under the MSFD, EU Member States are required to determine a set of characteristics for GES on the basis of eleven Quality Descriptors, each addressing a critical component of the ocean ecosystem or a form of pertinent human impact. One of the eleven Quality Descriptors of the ocean ecosystem is seafloor integrity. According to the criteria outlined in the MSFD, the integrity of the seafloor must be "at a level that ensures the structure and functions of ecosystems are safeguarded and benthic ecosystems, in particular, are not adversely affected" (Barry, 2013).

The benthic environment is exploited as a natural resource by commercial fishing, hydrocarbon and aggregate extraction, aquaculture, marine renewable energy and tourism industries (Halpern et al., 2008, Halpern et al., 2015). It plays a vital role in the structure and functioning of marine ecosystems through the recycling of nutrients, provision of high levels of secondary production and the dispersal and burial of sediments (Snelgrove, 1998). Recent growth in human interaction with the marine environment has driven the expansion of applied ecological studies of the benthic environment, many of which are focused on variation in benthic macrofaunal distributions in relation to human impacts (Borja et al., 2000, Borja, 2002, Muxika et al., 2005, Kennedy et al., 2011, Forde et al., 2013, Forde et al., 2015). Extensive literature exists on benthic habitat status assessment within coastal and transitional waterbodies (Borja et al., 2000, Muxika et al., 2007, Borja et al., 2011, Kennedy et al., 2011, Forde et al., 2013). There is potential to apply these methodologies in offshore MSFD mapping exercises in relation to putative disturbance sources (Borja et al., 2010, Barry et al., 2013).

The sedentary nature, or limited mobility of benthic infauna prevents their evasion of adverse conditions (Wass, 1967). The predictable, functional response of species to disturbance has resulted in them being placed into ecological groups according to their tolerance to stress (Pearson and Rosenberg, 1978, Glémarec and Hily, 1981, Grall and Glémarec, 1997). This qualitative weighting of species allows their distributions to be used as bioindications of disturbance in the benthic environment (Kennedy et al., 2011, Forde et al., 2013, O'Carroll et al., 2016, O’Carroll et al., 2017).

Ecological Quality Ratios (EQRs) were developed to assign Ecological Status (ES) to coastal and transitional waterbodies designated under the WFD. ES is assigned through the assessment of waterbodies based on biological, hydromorphological and physicochemical quality elements and the ultimate aim of the WFD is for all European waterbodies to have an ES of ‘Good’ or better. An EQR is derived by comparing putatively disturbed conditions to reference undisturbed conditions (Borja et al., 2000, Borja et al., 2007). EQRs have a decimal value between 0 and 1. Values close to 1 indicate a “High” ES and values close to 0 indicate an ES of “Bad”. There are five ES classes (Bad, Poor, Moderate, Good, High,) each represents a range of values that exist between class boundary cut off points along the full EQR value range. For the purposes of the WFD, EQRs must include metrics that address ‘the level of diversity and abundance of invertebrate taxa’ and ‘the proportion of disturbance sensitive taxa’ (Borja et al., 2007). The successional model developed by Pearson and Rosenberg (1978) is central to many benthic indices used in soft subtidal sediments (Borja et al., 2000, Prior, 2004, Borja et al., 2007, Muxika et al., 2007, Mackie, 2009, Phillips et al., 2014). The “Moderate”/ “Good” boundary is critical under the WFD. If an area is assigned an ES of Moderate or less, remedial management action is required to improve the ES of the area to Good or higher. The use of EQRs in baseline and monitoring surveys of construction or extraction processes will give information on the 'status' of a habitat that is more robust against seasonal and interannual variability than standard data analyses (Reiss and Kröncke, 2005, Kröncke and Reiss, 2010).

The 'Bottom-Up' habitat mapping approach produces spatial units that are delineated by boundaries which represent groups of stations that have similar faunal compositions (Conor et al., 2004). This approach is particularly useful for detecting human impacts on the benthic environment (Pearson and Rosenberg, 1978, Underwood and Peterson, 1988, Underwood, 1991, Borja et al., 2000). However, this approach is labour and cost intensive and not logistically feasible as a habitat mapping method in large scale surveys of the offshore environment.

Recent advances in remote sensing technologies have encouraged the use of the "Top-Down" monitoring approach (Kostylev et al., 2001, Christensen et al., 2009, McGonigle et al., 2009, Brown et al., 2011). This approach holds the assumption that distinct topographic features will host distinct biological assemblages (LaFrance et al., 2014). Biological data taken from comparably small proportions of these topographic features (Brown et al., 2002, Solan et al., 2003, Eastwood et al., 2006) are then extrapolated across the feature area so that a biological characterization can be produced.

There is an increasing need for response (Dalleau et al., 2010) and predictor (Brown et al., 2011) variable surrogates in large scale mapping exercises. The premise of adopting surrogates for data types that are logistically challenging to acquire, is routed in the fact that large scale ocean conservation initiatives must adopt time and cost effective data acquisition techniques. High resolution sediment grain size data is frequently classified as coarse resolution nominal descriptors when reporting on the conservation status of sedimentary habitats. Studies carried out in designated Natura 2000 sites commonly use the EUNIS sediment classification system (*sensu* Long, 2006) when reporting on the status of sedimentary habitats (Kennedy et al., 2008). Recent developments in acoustic mapping technologies present new time effective methods of seafloor data acquisition over large spatial scales (Brown et al., 2011). Acoustic signal derivatives have been successfully used as surrogates for sediment particle size analysis (PSA) outputs in large scale benthic habitat mapping studies (Ehrhold et al., 2006, Cook et al., 2008, Dolan et al., 2008, Brown and Blondel, 2009, Buhl-Mortensen et al., 2009, Callaway et al., 2009, Dolan et al., 2009, Brown et al., 2011). Issues such as fine scale sediment heterogeneity and gradational changes in sediment characteristics can affect acoustic derivative representativeness at small spatial scales (Brown and Collier, 2008). However, these issues become less pertinent with increasing spatial scale, or within areas of homogenous habitat (Monteys et al., 2016).

In large areas that incorporate multiple physical habitat types, abiotic factors such as ocean currents structure benthic macrofaunal distributions through food provision (Tweedle, 2005, White et al., 2005, Mienis et al., 2007), the distribution of larvae (Diesing et al., 2009, Cowen and Sponaugle, 2009) and through influencing the seafloor substrate characteristics (Posey and Ambrose Jr, 1994, Dos Santos Brasil and Goncalves da Silva, 2000, Sarkar, 2000). At local habitat scales the interaction of currents with benthic macrofauna are highly variable and dependent on the ecological resolution observed (Okamura, 1984, Okamura, 1985, Okamura, 1988, Underwood and Petraitis, 1993, Vogel, 1994, Vogel, 1996, Underwood and Chapman, 1996, Wootton, 2001, Terlizzi et al., 2007). Studies that have effectively used hydrodynamic parameters to explain spatial variation in benthic macrofaunal distributions have focused on coarse spatial resolutions (Christensen et al., 2009, Guinan et al., 2009a, Dutertre et al., 2013, Rengstorf et al., 2014). The ability of hydrodynamic models to explain variability in benthic community structure over large areas could make them useful tools for large scale mapping studies required to meet the criteria of the MSFD.

One of the issues facing EU member states with large EEZs is the cost associated with the extensive mapping of the benthic environment. The bottom-up mapping approach is prohibitively costly at large spatial scales, yet the top-down procedure may require significant investment in hydrodynamic model development and large scale acoustic seabed classification surveys. The Republic of Ireland is well equipped in this regard (Diesing et al., 2009). A significant proportion (125,00km^2) of Irelands EEZ has been acoustically mapped and classified by the Irish National

Seabed Survey (now INFOMAR: Integrated Mapping for the Sustainable Development of Irelands Marine Resources).

Using publically available data sources in conjunction with time-efficient survey techniques could significantly reduce costs associated with large scale surveys. Sub sampling sediments from single grab samples has been shown to be an time effective survey technique (Somerfield and Clarke, 1997). The limitations of this technique are more pertinent when applied to coarse sediments, and when used in high resolution habitat quality assessments (Kennedy et al., 2011). This must be taken into consideration during survey planning stages. Cost effective survey designs are integral to maximising survey outputs within budgetary limits (Clements et al., 2010), however, it is important to ensure survey designs do not affect the integrity of the outputs (de Jonge et al., 2006).

Artificial Neural Networks are computational algorithms that learn from experience in a way that is very similar to the animal brain (Lek and Guégan, 1999). ANNs are used in a variety of applications from speech (Chu and Bose, 1998) and image recognition (DeKruger and Hunt, 1994) to the prediction of brain death rates in neurological intensive care wards (Liu et al., 2011). Since ANNs were first suggested as potentially useful tools in ecological modelling (Colasanti, 1991, Edwards and Morse, 1995) the applications in which they are applied have diversified rapidly (Baran et al., 1996, Brosse et al., 1999, Özesmi and Özesmi, 1999, Chon et al., 2000, Wei et al., 2001, Wilson and Recknagel, 2001, Bradshaw et al., 2002, Lee et al., 2003, Joo et al., 2011, Watts et al., 2011, Song et al., 2013, Awad, 2014, Coad et al., 2014, Santos et al., 2014). ANNs have been shown to perform significantly better than multiple linear regression models at elucidating non-linear relationships (Baran et al., 1996, Brosse et al., 1999). ANNs have been applied to model ecological processes at multiple scales, from the microbial (Santos et al., 2014), to the geographical distribution of fur seal breeding grounds (Bradshaw et al., 2002), to coastal algal blooms (Wei et al., 2001, Lee et al., 2003) and broad scale predictive mapping of rocky reef habitats (Watts et al., 2011).

This study aims to develop a predictive model for the spatial distribution of EUNIS biotope classifications in Galway Bay, Ireland. This modelling approach was developed with the aim of making it applicable in offshore, MSFD benthic mapping exercises, similar to other previous studies (Coltman et al., 2008, Degraer et al., 2008). Predictive models such as this will be required to form baseline biological maps of Irelands EEZ under the MSFD, as it is not logistically feasible to physically sample the entire Irish seascape. In this study, time efficient survey techniques were adopted to test their viability in large spatial scale benthic habitat mapping. As much publically available data as possible was incorporated in our model which included depth and acoustic seabed classifications, transitional water boundaries and grab survey data. The inclusion of widely available data is recommended to maximise the applicability of a predictive spatial model (Degraer et al., 2008). The hydrodynamic model was developed in-house and is a high resolution 3D numerical model of Galway Bay (the Environmental Fluid Dynamics Code; Ren et al., 2015). We also demonstrate the use of a benthic EQR, the Infaunal Quality Index (IQI: Phillips et al., 2014), as a tool for monitoring a water body in relation to a point source of human disturbance. We apply the IQI to the data to assess the ES of Galway Bay.

2. Materials and Methods

2.1 Study area

Galway Bay is a large semi enclosed Bay on the West of Ireland (See Figure 1). The opening to the Bay faces westwards into the NE Atlantic Ocean. Galway Bay is relatively sheltered from large wave activity due to the alignment of the three Islands (Inis Mor, Inis Meain and Inis Beag) along a NW-SE axis across the Bays' opening (See Fig. 1). The islands are separated from each other by Gregory and Foul sounds, and from the mainland by North and South sounds. These four channels link the Bay with the Atlantic. East of Black Head is shallow with depths less than 30m, the outer Bay is up to 70m deep near the North Sound. Galway Bay exhibits a semidiurnal tidal regime with a range of approximately 5m. Water currents within the Bay are counter-clockwise, entering from the South Sound between Blackhead and Inisheer and exiting through the North Sound between Slyne Head and Inishmor.

Inner Galway Bay is designated as a Natura 2000 site, the Galway Bay Complex (SAC 000268), due to the multiple qualifying interests that it contains. Under Annex I of the HD the inner Bay qualifies as 'large shallow inlets and bays' (Qualifying interest 1170). Within the inner Bay there are significant 'mudflats and sandflats not covered by seawater at low tide' (Qualifying interest: 1140), and widespread intertidal and subtidal 'reefs' (Qualifying interest: 1170). The inner bay is also designated as an SPA (The Inner Galway Bay SPA 004031) under the BD as it contains important feeding and breeding grounds for native and migratory birds.

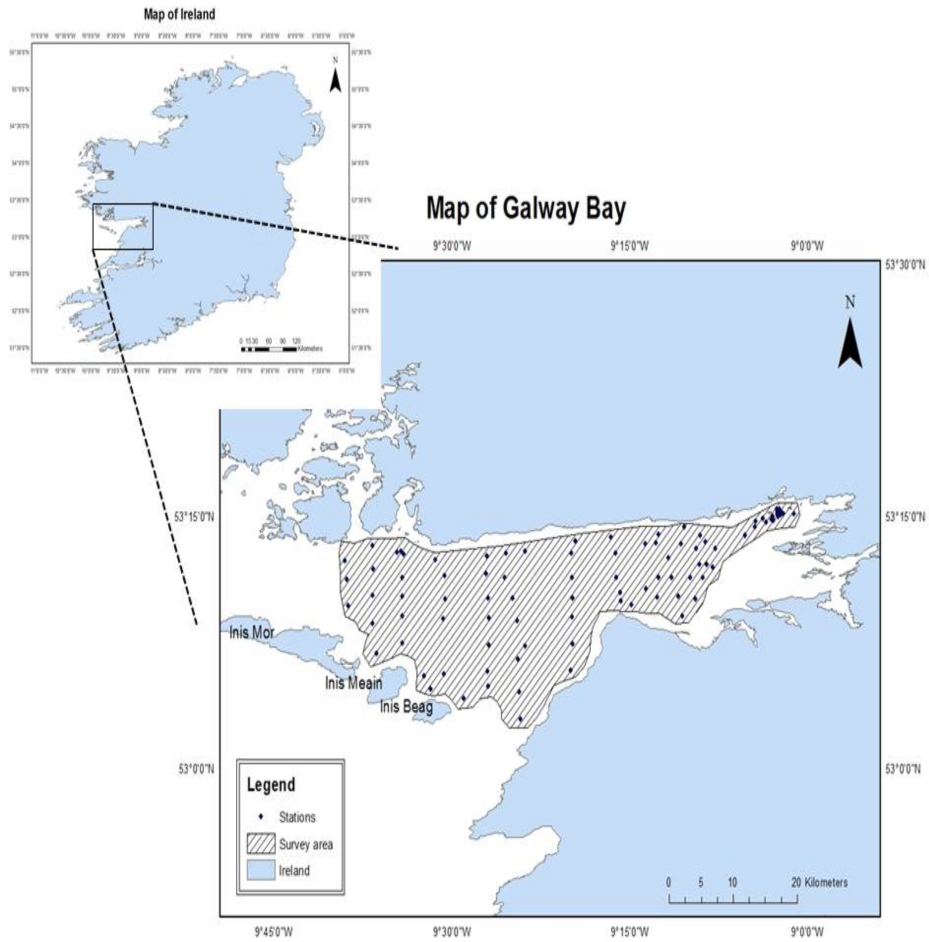


Figure 1: Map of Ireland (top left) and a map of the survey area (Galway Bay) including station locations.

2.1 Field Surveys

Data for this study was gathered from multiple sources. The biological data used in statistical analyses was collected in: *July 2004*; publically available data collected as part of the environmental impact statement by independent consultants for a harbour extension (8 stations: http://www.galwayharbourextension.com/), *August 2009*; publically available data gathered as part of a Natura 2000 grab survey of Galway Bay by independent consultants (2 grab stations: https://www.npws.ie/faq/maps-and-data), *February 2010*; publically available data from a follow up grab survey for the dock extension environmental impact statement (5 grab stations: http://www.galwayharbourextension.com/). *March 2011*; a training cruise for undergraduate students within Galway Bay (41 grab stations, taxonomic analyses were carried out by the corresponding author). *September 2014*; a grab survey carried out specifically to complete spatial coverage of the survey area (36 stations, taxonomic analyses were carried out by the corresponding author). Taxonomic analyses completed on all publically gathered data was carried out by independent consultants.

2.2 Sample processing

In the 2004 and 2010 grab surveys, two replicate Van Veen grab samples were used as faunal samples at each station. Separate grab samples were taken for sediment particle size analysis. The 2009, 2011 and 2014 surveys took one Day grab sample per station, and subsampled each grab for sediments. In all cases, samples were sieved on a 1mm mesh sieve, fixed with 10% buffered formalin and preserved in 70% alcohol. The taxa were then identified to species level where possible.

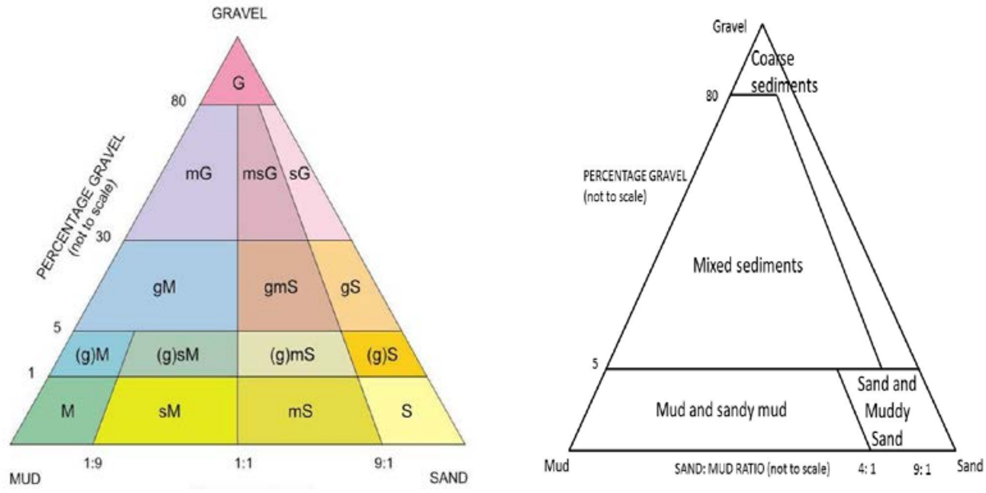
2.3 Sediment data processing

Associated sediment data for each faunal sample was derived from textural groups produced by GRADISTAT (Blott and Pye, 2001) in the case of public data sources (2004, 2009, 2010 surveys), and from field descriptions in the case of in-house surveys (2011, 2014). Textural group sediment classifications correspond to fine scale classifications of the Folk and Ward (1957) classification method (See Figure 2A). Textural group classifications were then converted into EUNIS classifications. EUNIS classifications correspond to coarse scale Folk and Ward (1957) classification *sensu* Long (2006) (See Figure 2B). EUNIS sediment classifications will hereinafter be referred to as ‘observed’ sediment classifications.

INFOMAR acoustic classifications were downloaded as a shapefile from the Seabed Viewer on the Irish Marine Institute’s website (http://maps.marine.ie/infomar/). In Galway Bay, INFOMAR acoustic classifications are split into four bottom types; mud to fine sands, fine sands to medium sands, coarse sands to gravel and reef. This classification is based on the mean sediment type. Each class is either finer or coarser than adjacent classes. There is no mixed sediment classification. INFOMAR sediment classifications will hereinafter be referred to as 'acoustic' sediment classes. Acoustic sediment classifications were produced using the QTC Multiview 3.0 software.

Textural groups were reclassified to correspond to acoustic sediment classes in two ways. In the first, all textural groups with >5% gravel were classified as coarse sand to gravel (See Figure 2 C). In the second, all textural groups with >30% gravel were classified as coarse sand to gravel (See Figure 2 D). The classification agreement between reclassified textural group classes and the acoustic sediment classifications was assessed using kappa analysis in SPSS 20 (IBM Corp., 2013).

(A) (B)



(C) (D)

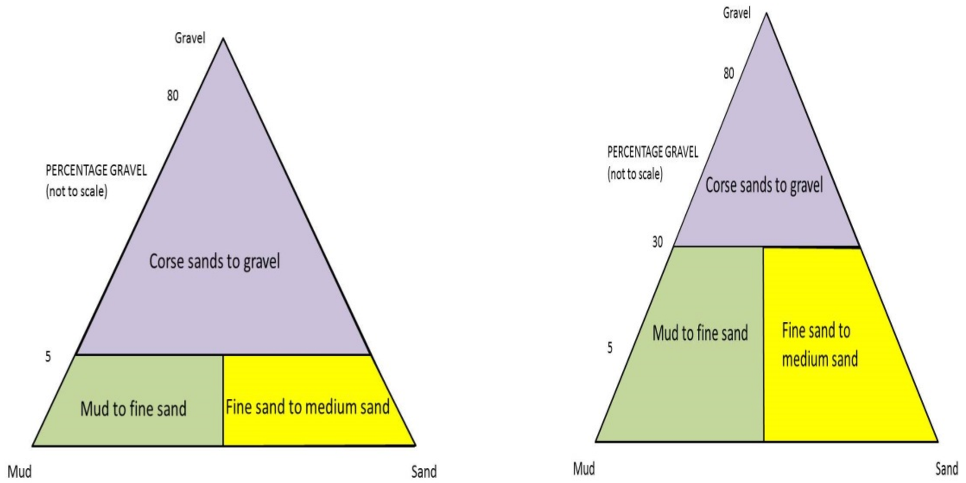


Figure 2: (A) Folk and Ward (1957) ternary classification triangle showing Textural Groups. (B) Simplified Folk and Ward (1957) triangle showing EUNIS sediment classes *sensu* Long et al (2006). (C) Triangle showing how textural groups were classified as acoustic sediment (INFOMAR) classes along the 5% gravel boundary. (D) Triangle showing how textural group classifications were classified as acoustic sediment (INFOMAR) classes along the 30% gravel boundary. **M**= Mud, **sM**= Sandy mud, **(g)M**= Slightly gravelly mud, **(g)sM**= Slightly gravelly sandy mud, **gM**= Gravelly mud, **S**= Sand, **mS**= Muddy sand, **(g)S**= Slightly gravelly sand, **(g)mS**= Slightly gravelly muddy sand, **gmS**= Gravelly muddy sand, **gS**= Gravelly sand, **G**= Gravel, **mG**= Muddy gravel, **msG**= Muddy sandy gravel, **sG**= Sandy gravel.

2.4 Hydrodynamic model

The numerical model Environmental Fluid Dynamic Codes (EFDC) was used to simulate the hydrodynamic circulation of Galway Bay. EFDC was developed at the

Virginia Institute of Marine Science by the U.S. Environmental Protection Agency (EPA). It comprises four linked modules: hydrodynamic, water quality and eutrophication, sediment transport, and toxic chemical transport and fate; only the hydrodynamic module was used for this research. This module solves the three-dimensional, vertically hydrostatic, free surface, turbulent averaged equations of motions for a variable density fluid. The hydrodynamic component of EFDC uses a semi-implicit, conservative finite volume solution scheme for the hydrostatic primitive equations with either two or three level time stepping (Hamrick, 2006, Hamrick, 2007, Hamrick, 1992). The model uses a sigma vertical coordinate system and either regular, or curvilinear orthogonal horizontal coordinates. The model has been applied to a variety of modelling studies of rivers, lakes, estuaries and coastal regions (Jin and Ji, 2004, Zou et al., 2006, O'Donncha et al., 2013). In this research, a model of Galway Bay was developed using a regular grid coordinate system. A 150 m horizontal spatial resolution was employed yielding 380×241 grid cells. Variable vertical layer thicknesses were used in the model. Thin layers were at the top and bottom of the water column with thicker layers in the middle. This ensured that wind forcing was not overly-damped by tidal forcing. Detailed description on setting up vertical layer structure for Galway Bay is included in Ren et al. (2015). The meteorological forcing data (wind, pressure, rain, solar radiation and relative humidity) were obtained at one minute intervals from the Informatics Research Unit for Sustainable Engineering (IRUSE) weather station located at the campus of National University of Ireland, Galway. Records of the River Corrib inflows, which enter Galway Bay were obtained from the Office of Public Works (OPW). Tidal water elevation time series generated from Oregon State University Tidal Inversion Software (OTIS) were used to define the tidal forcing on the western and southern open boundaries in the model (Egbert and Erofeeva, 2002, Padman, 2004).

2.5 Other data sources

Salinity data was downloaded from the Environmental Protection agency map viewer webpage (http://gis.epa.ie/Envision/). The data was downloaded as a transitional water boundary for the survey area and was converted into a binary predictor variable. Depth was also downloaded as a raster file from the Marine Institute website (http://maps.marine.ie/infomar/).

2.6 Statistical analyses

Environmental data was assigned to grab stations using the spatial join tool in ArcGIS. This tool joins adjacent environmental data to georeferenced grab stations by appending attributes to the grab stations' attribute table. Multivariate statistical analyses were carried out using PRIMER V6 (Clarke and Gorley, 2006, Clarke et al., 2006). A Bray-Curtis (BC) similarity matrix (Bray and Curtis, 1957, Clarke et al., 2006) was created using SACFOR transformed abundance data *sensu* Conor et al. (2004). Habitat classification at each station was performed in accordance with the 04-05 JNCC habitat classification scheme (Conor et al., 2004). Hierarchical clustering and Similarity Profile (SIMPROF) analyses (Clarke et al., 2008) were carried out on the Bray-Curtis resemblance matrix. SIMPROF analysis is a test of the null hypothesis of no significant group structure within the faunal matrix. It produces groups of samples that are significantly different to other groups within the dataset. SIMPER analysis (Clarke, 1993) was performed to determine the characterizing species for each group identified by SIMPROF. These data were compared to the biotope descriptions of Conor et al (2004) to determine which biotope the groups corresponded to.

Multinomial models were used to assess the ability of the environmental variables to predict the spatial distributions of EUNIS biotopes. Any biotopes represented by only one station were omitted from training datasets. Multinomial models were constructed using the nnet package (Venables and Ripley, 2002) in the R statistical software environment v3.3.1 (R Development Core Team, 2014).

Depth was used as a continuous variable. Depth averaged current velocity (ms^{-1}) (DAVG) was used as an ordinal predictor variable and contained three levels. The three levels of DAVG were chosen using jenks in ArcGIS. This method selects 'natural' groups of values based on the distribution of depth averaged velocity magnitudes along an X-axis. The class boundary cut off points were determined by selecting asymptotes in a trimodal distribution curve fitted to the distributions of the depth averaged velocity values observed. Salinity was used as a binary predictor variable. Proximity to reefs (Reef50) was used as a binary predictor variable. Acoustic sediment classifications were used as ordinal predictor variables and contained three levels. Observed sediment classifications were also used as ordinal predictor variables and contained four levels.

To test for the differences in the effectiveness of observed and acoustic sediment classifications as predictor variables, two identical multinomial models were run with one difference in each case. The first included acoustic classifications. The second included observed sediment classifications as sediment predictors. All ordinal predictor variables (DAVG, Depth, acoustic and observed sediment classifications) were normalised prior to multinomial modelling. The full dataset was used to train the multinomial models. This method can be used to assess the total predictive power of the entire dataset (Johnson and Omland, 2004), which can be useful in the case of small sample sizes. The multinomial modelling procedure identifies predictor variables as not significantly contributing to model outputs if $p = >0.05$. These predictors can then be removed from the model to improve the model AIC, F statistic or model fit. In some instances it is acceptable, and necessary to use model fit (i.e. number of correct predictions) as a selection parameter for variables as opposed to AIC or significance values (Zuur et al., 2007). In the current study, model fit was used as the selection criteria for environmental predictor variables. Optimal model comparison was done by comparing multinomial model likelihood ratio test (LRT) statistics. The likelihood ratio test (LRT) is one of the most commonly used approaches for model selection in ecology (Johnson and Omland, 2004). The significance of the difference in LRT values between models is determined by a Chi-Squared test (Zuur et al., 2007).

Two feed-forward neural networks were constructed using the nnet package (Venables and Ripley, 2002) in the R statistical software environment v3.3.1 (R Development Core Team, 2014). Both contained two hidden layers. There is no general consensus or established methodology for choosing the number of hidden layers for an ANN (Jha, 2007). However, ANNs with single or double hidden layers have been shown to be capable of carrying out complex analyses (Huang, 2003, Jha, 2007).

Each ANN used the set of environmental predictor variables deemed as optimal for making predictions by the multinomial modelling procedure. One ANN used acoustic classifications as the sediment predictor variable (acoustic sediment ANN), the other included observed sediment classifications as predictor variables (observed sediment ANN). Random stratified selection of stations was used to create the training dataset for both ANNs. Each ANN was trained with the same training dataset, only the sediment predictor variables were changed in each case.

Fleiss' Kappa for multiple raters (Fleiss, 1971, Conger, 1980, Fleiss et al., 2013) was carried out using the 'irr' statistical package (Gamer, 2015) in R (R Development Core Team, 2014) to assess the agreement between ANN predictions and the observed distributions of biotopes in the survey area.

2.7 Infaunal Quality Index

IQI was calculated for each faunal core using the IQI version 4 (freely available at: http://www.wfduk.org/resources%20/coastal-and-transitional-waters-benthic-invertebrate-fauna) of the propriety tool in Microsoft Excel developed by the UK Environment Agency (Phillips et al., 2014). The IQI calculation involves truncation of the species lists, spelling and synonym standardisation. The IQI EQR, a continuous variable between 0 and 1, is calculated by Equation 1.

$$IQI = \frac{\left(0.38 \times \left(\frac{(1-\mathrm{AMBI}/7)}{(1-\mathrm{AMBI}/7)_{\mathrm{ref}}}\right)\right) + \left(0.08 \times \left(\frac{(1-\lambda')}{(1-\lambda')_{\mathrm{ref}}}\right)\right) + \left(0.54 \times \left(\frac{\mathrm{S}^{0.1}}{\mathrm{S}^{0.1}{}_{\mathrm{ref}}}\right) - 0.4\right)}{0.6} \tag{1}$$

Where:

- AMBI is the AZTI Marine Biotic Index (Borja et al. 2000),
- 1–λ' is Simpson's evenness index,
- S is number of species,
- ref parameters are the maximum reference values for the habitat.

The IQI tool sets reference conditions for each component of the metric based on local environmental parameters including sediment particle distribution and salinity classification. Environmental parameters can be entered into the tool as continuous or categorical variables. For each core, salinity classification was entered as "coastal" while sediment parameters were entered as % weight values across full Phi classes ranging between 4 and -2 units. Within the IQI tool AMBI was calculated using the 2012 species list available on http://ambi.azti.es/. The AMBI value is a continuous variable based on the proportions of five ecological groups to which the species are allocated (Borja et al., 2000). IQI EQR values are converted to ES classes using the following class boundary values; Good–High, 0.75; Moderate–Good, 0.64; Poor–Moderate, 0.44; Bad–Poor, 0.24.

The "Moderate/Good" is deemed the critical boundary under the WFD with remedial management action required to restore areas classified as Moderate (or worse) to Good (or better).

3. Results

3.1 Benthic community characteristics

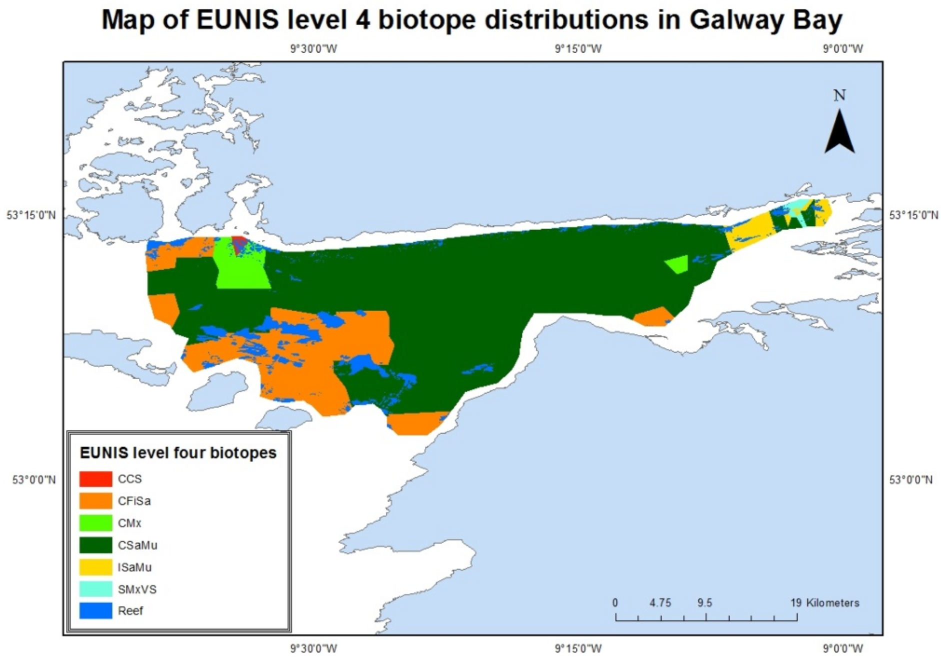


Figure 3: Map of the distributions of EUNIS level 4 biotopes in Galway bay. Note: Spatial units present in the map are derived from creating Thiessen polygons around each station location. The confidence of classification reduces with increased distance from sampling points.

Six EUNIS level 4 biotopes were found to exist within Galway Bay. One station lay on gravel within 50m of reef in the mouth of Rosaveal Harbour in the North Bay and was classified as 'Circalittoral Coarse Sediment' (SS.SCS.CCS). This biotope was dominated by the polychaetes *Spiophanes bombyx, Nephtys Hombergii* and *Sthenelais limicola*, the echinoderm *Echinocyamus pusillus,* and the venerid bivalve *Dosinia exoleta.*

The 'Circalittoral mixed sediment' (SS.SMx.CMx) biotope was found to exist in two discrete areas. In the North outer bay, three stations lay on mud, coarse, and mixed sediments surrounding the SS.SCS.CCS biotope in the mouth of Rosaveal Harbour. In the inner Bay, the biotope occurs at one station in the centre of the inner bay on mixed sediment which was largely sandy mud mixed with shell hash. The characterising fauna for this biotope were species of nemertean, polychaetes such as *Diplocirrus glaucus, Prionospio fallax, Podarkeopsis capensis* and species of Polynoidae, the bivalve *Chamelea striatula,* and the echinoderm *Amphiura filiformis.*

Five of the innermost stations that occurred in close proximity to the dredge channel were classified as 'Subtidal mixed sediments in varying salinity' (SS.SMx.SMxVS). This biotope lay on mixed sediments within the transitional boundary. This biotope was characterised by polychaetes of the genus Nephtys, others such as *Scoloplos armiger, Melinna palmata, Spiochaetopterus typicus,* and the bivalves *Tellina tenuis* and *Kurtiella bidentata.*

Adjacent to the SS.SMx.SMxVS biotope, within the transitional boundary, eight stations were classified as 'Infralittoral Sandy Mud' (SS.SMu.ISaMu). The most common sediment type was sand, but the biotope contained some stations that were classified as muddy, coarse and mixed sediments. This biotope was characterised by the bivalves *Thyasira flexuosa, Kurtiella bidentata,* the gammarid amphipod *Ampilesca brevicornis,* polychaetes of the genus Nephtys, and the tubiculous polychaetes *Euclymene oerstedi, Spiochaetopterus typicus*, and *Melinna palmata.*

Sixty stations were classified as 'Circaittoral sandy mud' (SS.SMu.CSaMu), forming the largest biotope in the survey area. This biotope occurs throughout the Bay, from the innermost infralittoral zone inside the transitional boundary, to the deepest parts of the circalittoral zone. 50% of stations in this biotope lay on mud, 8% lay on mixed sediments, and 40% lay on sand. This biotope was dominated by polychaetes such as *Hilbigneris gracilis, Nephtys hombergii, Spiophanes bombyx, Magelona alleni, Spiochaetopterus typicus,* and one bivalve dominant, *Phaxas pellucidus.*

The biotope 'Circalittoral fine sand' (SS.SSa.CFiSa) largely existed adjacent to, or in large patches of sediment between reefs in the outer bay. The faunal dominants for this biotope were mainly the echinoderm *Echinocyamus pusillus,* bivalves such as *Thracia phaseolina, Abra nitida, phaxas pellucidus* and *Dosinia exoleta* and polychaete *Spiophanes bombyx*.

3.2 Ecological Status of the benthic habitat in Galway Bay

The subtidal sedimentary environment of Galway Bay was found to have an ES of mostly 'High' with only one station being classified as 'Good'. As no stations were classified as 'Moderate' or less it was not deemed necessary to adjust the class boundary cut off points of the IQI to adjust for the potential effects of sediment subsampling on the number of species in some of the grab samples.

3.3 Sediment classification agreement

Kappa analysis of the agreement between the reclassified textural groups and the acoustic sediment classes found the agreement to be poor in both cases. The first method (using the 5% gravel boundary between coarse sands to gravel and the other classes) showed the poorest agreement with acoustic sediment classes (Kappa value= 0.078, p= 0.169). The second method (using the 30% gravel boundary between coarse sands to gravel and the other classes) showed slightly higher agreement with acoustic sediment classes (Kappa value= 0.1, p= 0.291). Neither agreement was statistically significant.

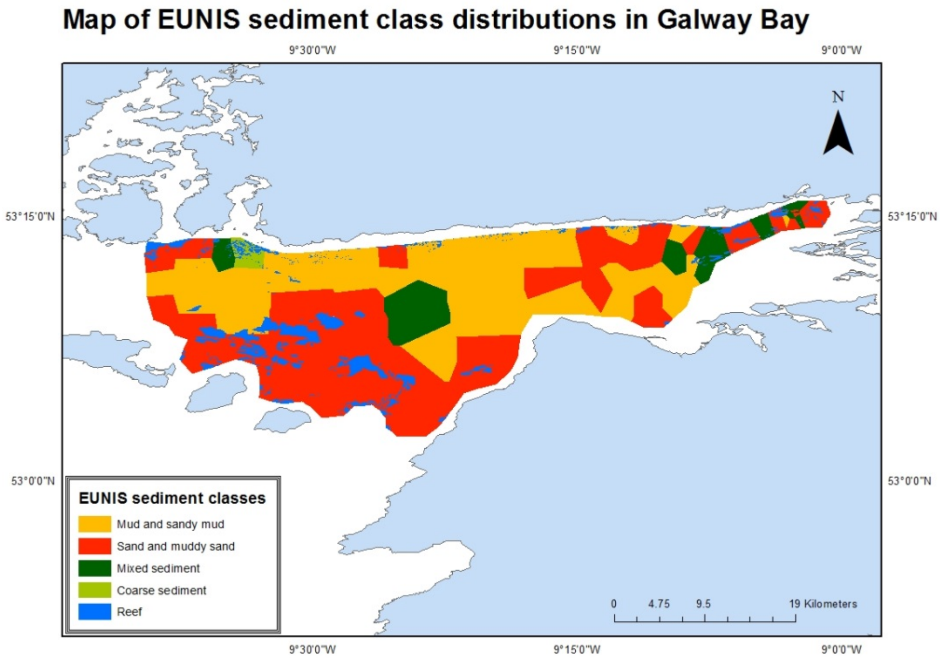


Figure 4: Map of observed (EUNIS) sediment class distributions in Galway Bay. Note: Spatial units present in the map are derived from creating Thiessen polygons around each station location. The confidence of classification reduces with increased distance from sampling points.

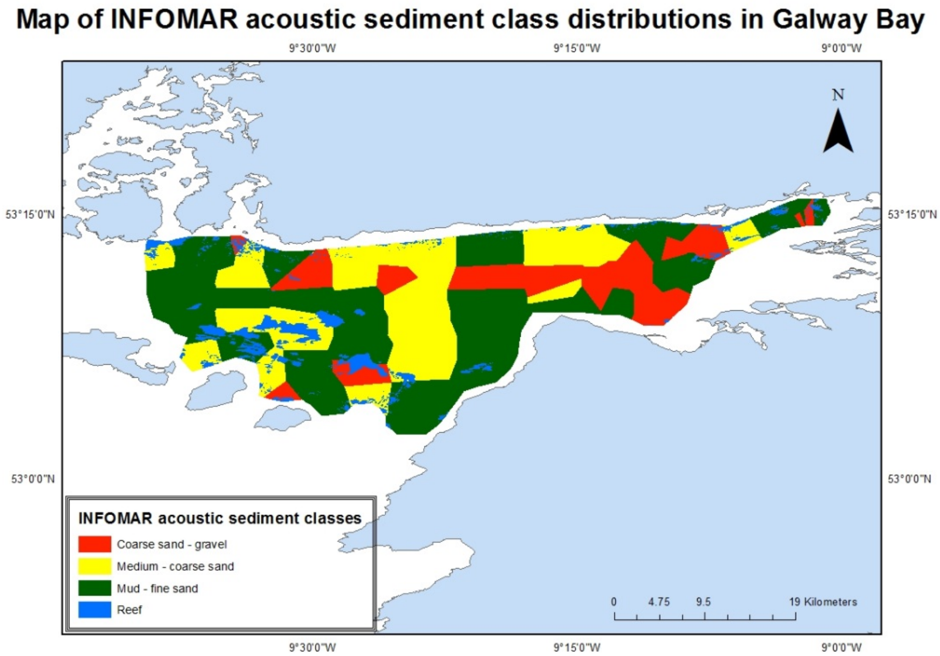


Figure 5: Map of acoustic (INFOMAR) sediment class distributions in Galway Bay. Note: Spatial units present in the map are derived from creating Thiessen polygons around each station location. The confidence of classification reduces with increased distance from sampling points.

3.4 Predicting spatial distributions of Biotopes

Both multinomial models could produce high rates of reclassification for EUNIS biotopes in Galway Bay (>70%). The multinomial model that used observed sediment classifications was significantly better than the multinomial model that used acoustic sediment classifications (See Table 1 E). This significant difference only amounts to a 7% difference in total correct model prediction rates (See Tables 1 E and F).

The optimal multinomial models in both cases use a combination of a sediment predictor variable (acoustic or observed), depth, salinity and proximity to reef (See Tables 1 A and B). Some non-significant predictor variables were retained in the optimal models (See Tables 1 A and B). These predictor variables were not removed as their removal reduced the fit of the model. Depth averaged current velocities did not contribute to model fit in either case and was not retained in the optimal models.

The predictions made by ANNs (See Supplementary material for perceptron diagrams and weightings) using the optimal models were approximately 10% lower than the reclassification rate of multinomial models (See Table 2). This is due to the reduced size of the training dataset used for the ANNs. The observed sediment ANN could predict biotopes 67% of the time and its predictions showed the highest agreement with observed biotope distributions (Kappa value= 0.34, p <0.001). The acoustic sediment ANN could predict biotopes 63% of the time and showed poorer agreement with observed biotopes (Kappa value= 0.24, p <0.0252).

Both multinomial models and ANNs performed well when making predictions on CSaMu (See Figure 3 for biotope distributions). For both multinomial models and ANNS, the observed sediment model predictions were more evenly distributed across the biotopes. Both multinomial models performed poorly when making predictions on CFiSa, despite this biotope occurring exclusively on sand in a discrete patch in the outer bay. The observed sediment ANN could make reasonable amounts of correct predictions for the CFiSa biotope when compared to the acoustic sediment ANN (See Table 2 C and D).

Multinomial models and ANNs that used acoustic sediment classes performed poorly when making predictions on mixed sediment biotopes. The CMx biotope included stations that occurred on coarse (1 station), mixed (2 stations) and sandy sediments (1 station) (See Figure 4 for observed (EUNIS) sediment distributions). Acoustic sediment classifications assigned all the sediments for this biotope as mud to fine sands or fine to medium sands (See Figure 5). Similarly the MxVS biotope occurred exclusively on mixed sediments, yet acoustic sediment classifications called the sediments muds to fine sands and fine to medium sands. The sediments of the inner biotope ISaMu were variable (Mx; 2 stations, Sa; 4 stations. Mu; 1 station) yet acoustic sediment classifications called them mud to fine sands.

Table 1: Results of multinomial models. (A) Optimal model that used acoustic sediment classifications. (B) Optimal model that used observed sediment classifications. (C) Goodness of fit table for the model that used acoustic sediment classifications. (D) Goodness of fit table for the model that used observed sediment classifications. (E) Performance table for the model that used acoustic sediment classifications. (F) Performance table for the model that used observed sediment classifications. (G) One-way Anova table comparing the optimal observed and acoustic multinomial models. **Df**= Degrees of freedom. **LRT**= L ratio test. **Pr(Chi)**= Probability produced by Chi Squared test. **RD=** Residual deviance. **1**= SS.SMu.CsaMu, **2**= SS.SMu.ISaMu, **3=** SS.SMx.CMx, **4**= SS.SMx.SMxVS, **5**= SS.SSa.CFiSa.

(A)	Df	AIC	LRT	Pr(Chi)
Intercept	164			
Depth	4	175	19.47	0.001
Salinity	4	158	2.12	0.714
INFO	8	154	6.93	0.544
Reef50	4	169	13.32	0.009
RD = 115.5				
AIC = 163.5				

(B)	Df	AIC	LRT	Pr(Chi)
Intercept	128			
Depth	4	138	17.9	0.001
Salinity	4	120	0.8	0.943
Reef50	4	126	6.7	0.155
EUNIS	12	154	50.8	1.00E-06
RD = 71.65				
AIC= 127.6				

(C)	Fitted				
Given	**1**	**2**	**3**	**4**	**5**
1	58	1	0	1	0
2	2	4	0	2	0
3	2	0	1	0	1
4	1	1	0	3	0
5	12	0	1	0	1

(D)	Fitted				
Given	**1**	**2**	**3**	**4**	**5**
1	57	1	0	0	2
2	2	6	0	0	0
3	2	0	2	0	0
4	0	0	0	5	0
5	11	0	0	0	3

(E)	No. of stations per biotope	No. of correctly predicted stns. per biotope	% correct predictions per biotope	Total correct predictions	Total % correct predictions
SS.SMu.CSaMu	60	58	96.6	67	73.6
SS.SMu.ISaMu	8	4	50		
SS.SMx.CMx	4	1	25		
SS.SMx.SMxVS	5	3	60		
SS.SSa.CFiSa	14	1	7.1		
Total stns.	91				

(F)	No. of stations per biotope	No. of correctly predicted stns. per biotope	% correct predictions per biotope	Total correct predictions	Total % correct predictions
SS.SMu.CSaMu	60	57	95	73	80.2
SS.SMu.ISaMu	8	6	75		
SS.SMx.CMx	4	2	50		
SS.SMx.SMxVS	5	5	100		
SS.SSa.CFiSa	14	3	21.4		
Total stns.	91				

(G)	Model	Resid. Dev	Df	LR stat.	Pr(Chi)
	INFO	115.48			
	EUNIS	71.65	4	43.84	6.94E-09

Table 2: Results of ANN predictions. (A) Goodness of fit table for the acoustic sediment ANN. (B) Goodness of fit for the observed sediment ANN. (C) Performance table for the acoustic sediment ANN. (D) Performance table for the observed sediment ANN. **1**= SS.SMu.CsaMu, **2**= SS.SMu.ISaMu, **3=** SS.SMx.CMx, **4**= SS.SMx.SMxVS, **5**= SS.SSa.CFiSa.

(A)	**Fitted**				
Given	**1**	**2**	**3**	**4**	**5**
1	26	1	0	0	3
2	2	2	0	0	0
3	2	0	0	0	0
4	1	2	0	0	0
5	5	0	1	0	1

(B)	**Fitted**				
Given	**1**	**2**	**3**	**4**	**5**
1	22	6	0	0	2
2	1	2	0	0	1
3	1	1	0	0	0
4	0	0	0	3	0
5	3	0	0	0	4

(C)	**No. of stations per biotope**	**No. of correctly predicted stns. per biotope**	**% correct predictions per biotope**	**Total correct predictions**	**Total % correct predictions**
SS.SMu.CSaMu	30	26	86.6	29	63
SS.SMu.ISaMu	4	2	50		
SS.SMx.CMx	2	0	0		
SS.SMx.SMxVS	3	0	0		
SS.SSa.CFiSa	7	1	14.3		
Total	46				

(D)	**No. of stations per biotope**	**No. of correctly predicted stns. per biotope**	**% correct predictions per biotope**	**Total correct predictions**	**Total % correct predictions**
SS.SMu.CSaMu	30	22	73. 3	31	67.4
SS.SMu.ISaMu	4	2	50		
SS.SMx.CMx	2	0	0		
SS.SMx.SMxVS	3	3	100		
SS.SSa.CFiSa	7	4	57.1		
Total	46				

4 Discussion

It was possible to make good levels of correct predictions on the distribution of EUNIS biotopes in Galway Bay using multinomial modelling and ANNs. Multinomial models trained with the entire dataset could make correct reclassifications approximately 75% of the time using both acoustic and observed sediment predictors. Predictions made by ANNs that were trained with half of the data made correct predictions on biotopes between 63% and 67% of the time. These results show the effect of small sample size on model performance, and indicate that a higher number of sampling stations could have enabled ANNs to perform better at making predictions (Hernandez et al., 2006).

Our entire dataset was comprised of different parts that were gathered at different times, by different operators, processed by different taxonomists, and during different seasons. Training a predictive model with a dataset that has such inherent variability should produce predictions that are robust against the possible confounding effects of seasonality and operator biases.

Multinomial models were principally used to assess which combination of environmental predictor variables form optimal predictive models. Carrying out iterative single term deletions indicated that including depth averaged current velocities did not improve model fit in any instance. The lack of explanatory power in depth averaged current velocity for the spatial distribution of biotopes is caused by the homogenous hydrodynamic regime in the area surveyed. In ecological terms, there was no significant variation in the hydrodynamic conditions.

Reefs were not surveyed as part of this study. It is possible that if biotope response variables for reef communities were included, hydrodynamic variables may have contributed more to our models. Hydrodynamic variables have been shown to significantly contribute to predictive models for benthic biotopes that make predictions over a range of seafloor morphologies within a survey area (Christensen et al., 2009, Dutertre et al., 2013). The area surveyed by Christiensen et al. (2009), in places, exhibited current velocities of up to 2.5 m/s. Within Galway Bay maximum current velocities peaked at 1.6 m/s, and this peak occurred over reef. Dutertre et al., (2013) surveyed an area that extended over

150km of coastline, and found hydrodynamic parameters to hold significant explanatory power for the spatial distributions of benthic communities. Hydrodynamic conditions were reported to be increasingly complex in close proximity to islands that existed within the survey area. Estuarine plumes also held significant influence over hydrodynamic conditions in large parts. Our results show that the effectiveness of hydrodynamic variables in explaining spatial variation in benthic communities is scale dependant, but also dependant on significant habitat heterogeneity existing in the area surveyed.

The multinomial models that used observed sediment classes as predictor variables performed consistently better than those using acoustic sediment classifications. The significant difference between the observed and acoustic sediment multinomial models only amounted to a 7% difference in total correct predictions. The observed sediment ANN also showed better agreement with observed biotope distributions and this agreement was highly significant. The difference in ANN correct predictions was only ~5% between the observed and acoustic ANNs. These are small margins of difference when considering the monetary savings that can be made through the use of publically available sediment data with significant spatial coverage.

Our results show that acoustic sediment classifications could be used as a surrogate for observed sediment classes when predicting the distributions of biotopes in Galway Bay, and that doing so will only incur a ~6% reduction in total correct predictions. This demonstrates the potential the INFOMAR data has for use in large scale benthic habitat mapping, which has major implications for legislative requirements of the MSFD. Other similar studies have also found acoustic signal derivatives to be effective environmental predictor variables for the spatial distribution of biotopes (Buhl-Mortensen et al., 2009, Ierodiaconou et al., 2011). This study adds to the growing literature which demonstrates the potential of acoustic sediment classifications for use in large spatial scale conservation efforts (Ehrhold et al., 2006, Cook et al., 2008, Dolan et al., 2008, Brown and Blondel, 2009, Buhl-Mortensen et al., 2009, Callaway et al., 2009, Dolan et al., 2009, Brown et al., 2011).

Within the innermost part of the Bay sediment type was variable and dominated by sands and mixed sediments. This variability is reflected in the spatial distributions of the ISaMu, MxVS and CSaMu biotopes in that area. INFOMAR acoustic sediment classifications classified the innermost Bay area as being largely mud to fine sands and fine sands to medium sands. This mismatch between observed and acoustic sediment classes could be a result of local heterogeneity in sediment distributions, which can cause acoustic swathe methods to miss fine resolution information on the seafloor (Brown and Collier, 2008, Brown et al., 2011). The lack of a mixed sediment class in the acoustic sediment classifications is likely to have been a factor in its poor performance when making predictions on the mixed sediment biotopes.

The local scale heterogeneity of biotope distributions in the innermost Bay area is also likely to have had an effect on the performance of multinomial models and ANNs that used acoustic sediment classifications. Classifications made on sediment subsampled from faunal grabs in the field will have a higher affinity for the biological community present (Somerfield and Clarke, 1997) compared to an acoustic classification that is based on the mean sediment type within an area. The affinity between field sediment observations and the faunal samples enabled the multinomial models and ANNs that used observed sediment classes to distribute their total correct predictions more evenly across the biotopes than the other acoustic sediment multinomial models and ANNs.

Heterogeneity in sediment distributions coupled with the intrinsically different procedures involved in deriving EUNIS or textural group sediment classes and INFOMAR acoustic sediment classifications had an effect on the agreement between the two. Textural groups are derived using the Folk and Ward (1957) ternary triangle. EUNIS sediment classes are derived through the use of a simplified Folk and Ward (1957) ternary classification triangle *sensu* Long (2006). This technique produces sediment classes that are based on grain size distribution ratios. The three INFOMAR acoustic sediment classes fit into grain size intervals; muds to fine sands (9-4 phi intervals or 2-63µm), fine sands to medium sands (4-1 phi intervals or 64-449µm) and coarse sands to gravel (1- (-1) phi intervals or 0.5-2mm).

The composition of shallow sediments such as those in the innermost Bay are temporally variable as a result of natural disturbance events such as sediment resuspension due to storm activity (Gray, 1981, O'Carroll et al., 2016). Acoustic sediment classifications may have been made at a time when sediments in the area were more homogenous, leading to the mismatch between observed and acoustic sediment classifications.

In the greater survey area the dominant biotope was CSaMu. Stations classified as CSaMu occurred on mud, sand and mixed sediments. This is a common occurrence when the bottom-up biotope classification method is used (Kennedy et al., 2008). The bottom-up procedure requires that groups are delineated based on the similarity of relative faunal abundances instead of physical environmental boundaries. In some instances, a stations' level four biotope classification did not agree with its' level three classification, which is based on sediment descriptions only (Top-Down approach). This can be caused by other factors such as salinity variation, organic content, water depth, aspect, exposure, current speed etc. having pronounced effects on macrofaunal communities either as main effects, or as interactions with grain size (Kennedy et al., 2008).

The CFiSa biotope dominated the outer fringes of the survey area and occurred exclusively on sandy sediments. Only the observed sediment ANN could make reasonable predictions (~60%) on the distribution of this biotope. Generally, when models misclassified this biotope, it was classified as the CSaMu biotope dominant. The CFiSa biotope only contained 14 stations compared to the 60 in the CSaMu. More sampling effort in the outer, more exposed sandy sediments would have allowed models to differentiate between the CFiSa and CSaMu biotopes more effectively (Hernandez et al., 2006).

Our results show that the classification of sediments as textural groups in the field using expert observation is an accurate and conveniently cost-effective method of sediment data collection. This method can be of benefit to benthic habitat mapping initiatives, such as those carried out by government agencies in response to EU Directives. Expert field classifications of sediments is another example of a cost effective methodology that can be used in mapping studies without affecting the integrity of the outputs (Barbour and Gerritsen, 1996, Somerfield and Clarke, 1997, de Jonge et al., 2006, Kennedy et al., 2011).

The good level of correct predictions made by the multinomial models and ANNs that used observed sediment classes suggests that subsampling sediment from grabs did not significantly affect sample affinities or the integrity of the faunal samples to any significant degree (Somerfield and Clarke, 1997). The motivation for single grab survey designs include; cost reduction, and the rationale that PSA distributions will have more affinity with macrofaunal distributions, allowing for their communities to be classified with increased confidence (Forde et al., 2012). This approach has been shown to significantly reduce the number of species in a sample. This has a negative effect on the ES assigned to a sample (Forde et al., 2012). Once this effect is taken into account prior to a monitoring study, it can be circumvented through the adjustment of class boundary cut off points of the EQR being applied to the data. The significant majority of stations in this study were classified as High, with only one station classified as Good. This meant that it was not necessary to adjust class boundary cut-off points to address the reduction of ES by subsampling sediments. The 'High' ES of the area surveyed is a result of the distance of the stations from the putative disturbance sources of the dredge disposal ground, dredge channel, and treated sewage outfall pipe (O'Reilly, 2006, Patterson, 2006).

Sub sampling sediments could also have an effect on the recording of rare or less numerous species. In the case of standard habitat mapping studies, sub sampling sediments has been found to slightly effect sample affinity with macrofaunal distributions, but not in an ecologically meaningful way (Somerfield and Clarke, 1997).

There was a general trend of coarse sediments existing within 50m of reefs in Galway Bay. This was reflected in the good performance of proximity to reef as a predictor variable in our models. Proximity to reef can have significant effects on sedimentary conditions and resident macrofaunal communities (Posey and Ambrose Jr, 1994). This effect can occur at spatial resolutions that can be detected by acoustic signal derivatives (Callaway et al., 2009). Our results, and those of others (Ierodiaconou et al., 2011, Callaway et al., 2009, Vaughn Barrie et al., 2011) have shown that it is worth considering proximity to reef as a predictor variable in spatial models.

Our results indicate that there is potential for acoustic sediment classifications to be used in large scale predictive mapping exercises. The factors contributing to the marginally poorer performance of multinomial models and ANNs that used acoustic sediment classifications are predominantly the result of local heterogeneity in sediment types within Galway Bay. The margin of difference in correct predictions between using either observed or acoustic sediment classifications is small, and arguably an acceptable margin of error when the potential cost savings of using acoustic sediment classifications are taken into account. The asymmetrical distribution of samples across biotopes resulted in some being under-represented in training datasets. This may also have affected model prediction rates (Hernandez et al., 2006). Confounding effects such as these can be expected to occur in heterogeneous habitats such as a large embayment. These effects are likely to be less pertinent in homogenous offshore habitats (Brown and Collier, 2008, Brown and Blondel, 2009, Brown et al., 2011, Monteys et al., 2016).

The methodology adopted for this study incorporated pre-existing datasets and cost effective field survey techniques. Our results indicate that this approach can be taken in other similar habitat mapping exercises. We have shown that acoustic sediment classifications have potential to be used in large scale mapping studies such as those required under the MSFD. We also present a time effective technique for seafloor sediment data acquisition using expert judgement in the field. This methodology could be useful for rapid ground-truth studies for predictions made on sediment distributions (Stephens and Diesing, 2015) in new areas. Further research is needed to assess how acoustic sediment classifications will perform in comparison to field sediment classifications in more homogeneous offshore habitats.

Regardless of sediment data acquisition methods, predictive spatial models will continue to play an important and growing role in the conservation of the ocean ecosystem (Buhl-Mortensen et al., 2009, Davies et al., 2008, Dolan et al., 2008, Guinan et al., 2009a, Guinan et al., 2009b, Dutertre et al., 2013, Rengstorf et al., 2013, Rengstorf et al., 2014), especially in relation to the sustainable management of marine resources (Council Directive 2008/56/EC). Co-ordinated seabed mapping initiatives such as INFOMAR are a useful asset for broad scale mapping exercises and Ireland is well equipped relative to many EU member states in this regard (Diesing et al., 2009). We have shown that it is possible to build a predictive model for EUNIS biotope distributions in a cost effective manner using available resources in combination with targeted data acquisition. The methodologies presented here have the potential to be applied in the offshore to meet Irelands' mapping obligations under the MSFD. Further research is needed to compare the effectiveness of acoustic versus observed sediment class based predictive models for the spatial distributions of EUNIS biotopes in offshore habitats.

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Supplementary material

Diagram of the multilayer perceptron from the EUNIS ANN

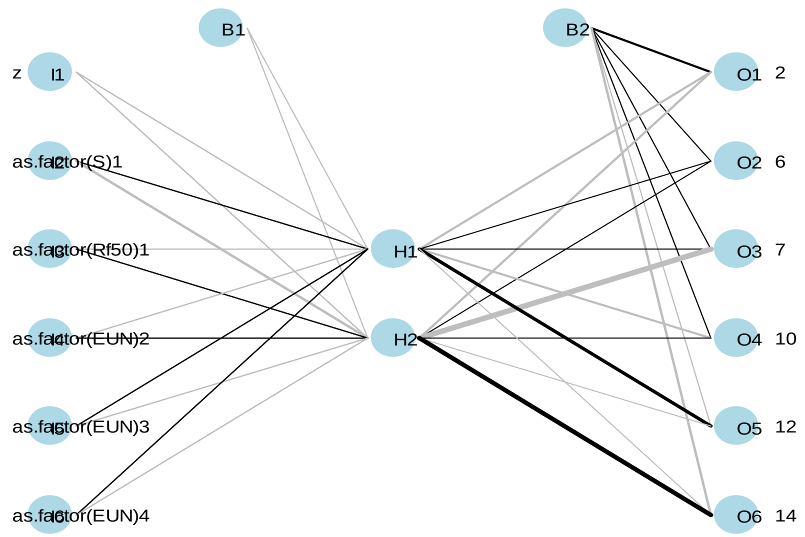

Figure a: Multilayer perceptron depicting the EUNIS feedforward ANN, its inputs (I1- I6), hidden layers (H1, H2), biases (B1, B2) and outputs (O1- O6). **Z**= depth (m), **S**= Salinity, **Rf50**= proximity to reef (within 50m), **EUN**= different levels of the factor EUNIS sediment class.

	Input to hidden layer weights		Biases to hidden layer weights		Biases to output weights		Hidden layer to output weights	
EUNIS ANN weightings	i1->h1	-0.01	b->h1	-0.7	b->o1	42.95	h1->o1	-55.7
	i2->h1	1.26	b->h2	-6.8	b->o2	17.46	h1->o2	14.63
	i3->h1	-13.2			b->o3	8.93	h1->o3	34.12
	i4->h1	-0.78			b->o4	22.9	h1->o4	-44
	i5->h1	1.35			b->o5	-31.38	h1->o5	85.6
	i6->h1	0.36			b->o6	-60.86	h1->o6	-34.6
	i1->h2	-0.39					h2->o1	-42.8
	i2->h2	-41.9					h2->o2	34.71
	i3->h2	8.65					h2->o3	-139
	i4->h2	1.63					h2->o4	34.69
	i5->h2	-14.1					h2->o5	-13.4
	i6->h2	-13.3					h2->o6	126.3

Table a: Weighted inputs, biases and outputs for each node of the ANN perceptron in Figure a.

Diagram of the multilayer perceptron from the EUNIS ANN

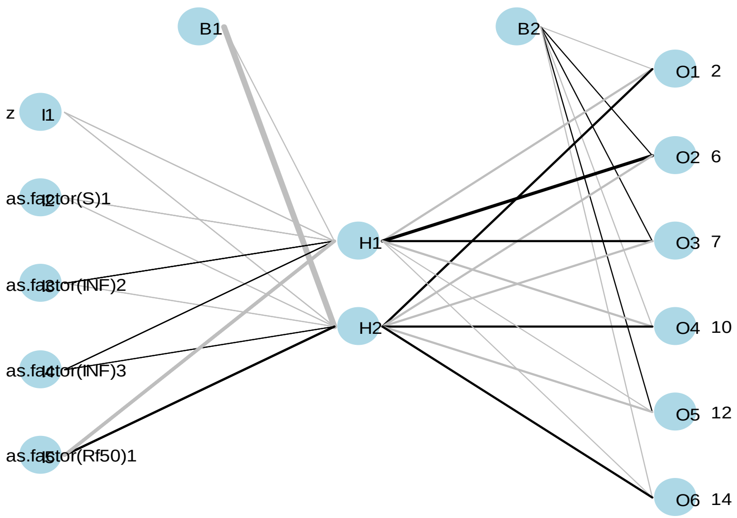

Figure b: Multilayer perceptron depicting the INFOMAR feedforward ANN, its inputs (I1- I6), hidden layers (H1, H2), biases (B1, B2) and outputs (O1- O6). **Z**= depth (m), **S**= Salinity, **Rf50**= proximity to reef (within 50m), **INF**= different levels of the factor INFOMAR sediment class.

	Input to hidden layer weights		Biases to hidden layer weights		Biases to output weights		Hidden layer to output weights	
INFOMAR ANN weightings	i1->h1	-0.17	b->h1	-2.11	b->o1	-1.34	h1->o1	-52.61
	i2->h1	-8	b->h2	-109.33	b->o2	1.36	h1->o2	67.03
	i3->h1	0.03			b->o3	5.41	h1->o3	54.55
	i4->h1	19.49			b->o4	-2.17	h1->o4	-27.51
	i5->h1	-60.91			b->o5	5.04	h1->o5	-25.08
	i1->h2	-3.88			b->o6	-8.3	h1->o6	-16.42
	i2->h2	-8.91					h2->o1	39.8
	i3->h2	-1.33					h2->o2	-44.56
	i4->h2	22.24					h2->o3	-52.06
	i5->h2	31.42					h2->o4	41.32
							h2->o5	-31.23
							h2->o6	46.76

Table b: Weighted inputs, biases and outputs for each node of the ANN perceptron in Figure b.