Intertidal Collection within the Berwickshire and North Northumberland Coast European Marine Site: investigating the scale, locale, and ecological impacts of harvesting *Arenicola marina*, *Arenicola defodiens*, and *Littorina littorea*

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Abstract

Robust evidence of fisheries impacts, fishing intensity, and spatial distribution of fishers are required, driven by a push towards evidence based management, and the trend towards Marine Spatial Planning (MSP). Intertidal fisheries have received considerably less research and management attention to date compared to inshore and offshore counterparts. The need for additional intertidal fisheries data, specifically within European Marine Sites (EMS), has been identified. This research focusses on the collection of lugworms *Arenicola marina* and *Arenicola defodiens,* and periwinkle *Littorina littorea* within the Berwickshire and North Northumberland Coast European Marine Site (BNNC EMS), UK. This thesis aims to provide an interdisciplinary evidence base for marine managers and future research to build upon.

Comparisons of sites experiencing a gradient of fishing pressure at the EMS scale, combined with small scale experimental disturbances, revealed the potential and actual impacts of local harvesting regimes. Data on the target species revealed no significant impacts between sites, suggesting that at current collection intensities, Northumberland populations of neither periwinkle nor lugworm are reduced or altered by fishing beyond naturally occurring levels. Community assessments revealed no observable impacts on the rocky shore, but sediment communities were negatively impacted with reductions in infaunal abundance and taxonomic richness, and altered community structure observed between sites and treatments. Recovery timescales were investigated and discussed.

Fisher distribution was mapped from shore observations, highlighting collection hotspots, and combined with questionnaire data to estimate biomass removal, with economic value discussed. Adherence to current fisheries regulations were

investigated, revealing a shortfall in existing enforcement measures, with illegal night time collection especially prevalent at some sites. Commercial and recreational collection characteristics were contrasted, and identification features recommended. Finally, spatial models of habitat suitability, sensitivity, and vulnerability were produced for the lugworm fishery, assessing the appropriateness of current spatial management measures. The spatial extent of existing bait digging byelaws included most of the highly vulnerable areas identified in the model outputs, with suggestions to further improve the coverage discussed.

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Chapter 1 : Introduction

1.1 Introduction to the Thesis

1.1.1 Background and Rationale

Coastal marine ecosystems are some of the most valued and productive habitats in the world (Costanza *et al.*, 1997). However they are often the most degraded, with ever increasing human pressures (Reid *et al.*, 2005). The impacts of activities and both potential and observed degradation of coastal ecosystems has gained more attention in recent years, and marine ecologists, managers, users, and policy makers are concerned about how they can be protected (Crain *et al.*, 2009).

The conservation of coastal ecosystems is both globally and locally important. At a global scale, The Convention on Biological Diversity has set an international target to protect 10% of coastal and marine areas by 2020, through designation of protected areas (Boonzaier and Pauly, 2016). This global network is made up of local and national networks, such as those within Europe and the UK. On a European scale there are areas designated as Special Area of Conservation (SAC) and Special Protection Area (SPA). SACs and SPAs with a 'marine area' (any land covered continuously or intermittently by tidal waters or any part of the sea in or adjacent to Great Britain up to the seaward limit of territorial waters) can be considered a European Marine Site (EMS) (The Conservation (Natural Habitats, &c.) Regulations, 1994). EMSs contribute to the global aim of protecting the oceans and coasts. The Berwickshire and North Northumberland Coast European Marine Site (BNNC EMS) is one example of these protected areas within the UK, and is the study site selected for this thesis, due to the availability of numerous coastal habitats subject to multiple human pressures, combined with the relevant legislation and management requirements to drive the research (MMO, 2014b).

The numerous anthropogenic stressors threatening coastal ecosystems include: habitat loss, climate change, eutrophication, pollution, invasive species, and overexploitation (Kay and Alder, 1998; Beatley *et al.*, 2002; Lotze *et al.*, 2006; Crain *et al.*, 2009). This thesis focusses on the threat from over exploitation. The intertidal zone is usually accessible over the tidal regime, and regularly exploited by humans. Intertidal exploitation has been occurring since prehistoric times (e.g. Thompson *et al.*, 2002; Erlandson *et al.*, 2011; Braje *et al.*, 2012), now with many organisms collected from the intertidal zone for both food and fishing bait, both recreationally

and commercially (Fowler, 1999). Both rocky shores and sand/mud flats in the intertidal zone are exploited in this way in Britain.

Ecologists have long pursued accurate assessments of the environmental impacts of multiple human activities, with the ultimate aim of protecting ecosystems from degradation. However, significant uncertainties remain, especially at local scales, as many activities and geographic locations remain poorly covered in the literature due to the large scale and distribution of human pressures. Little research into intertidal collection activities has been conducted in the North-East of England, and management concerns over currently unidentified impacts mean that there is a real need for research within the BNNC EMS. The main driver of this thesis is Department for Environment, Food & Rural Affairs (DEFRA) 'Revised Approach to the Management of Commercial Fisheries in European Marine Sites', which was announced in August 2012. This project is now known as the 'Fishing in MPAs' project. Information on anthropogenic activities is needed to inform effective management. Within protected areas, such as EMSs, fishing activities are only allowed if they do not undermine the conservation objectives of the site, or impact upon the site integrity (MMO, 2014b). Therefore, every fishing activity occurring within an EMS must undergo a Habitat Regulation Assessment (HRA) in agreement with Article 6 of the Habitats Directive (Council Directive 92/43/EEC), with the aim of assessing possible impacts on the site's designated features. If it is deemed possible or likely that a significant impact could occur from an activity, an appropriate assessment must be completed. This assessment will also inform management options to ensure the maintenance of site integrity. An evidence gap was acknowledged for intertidal collection activities for both rocky and sediment shores (assigned an amber rating – meaning the impacts are unknown), which is required to be filled before informed management decisions can be made.

Many species are collected throughout the BNNC EMS. Rocky shores are generally used for the collection of crabs, periwinkles, and mussels, whilst sandy shores are used for the collection of worms and crabs (Fowler, 1999). However, this study focuses on three target species, as a study on all collected species is beyond the scope of this thesis. The species investigated are the lugworms *Arenicola marina* and *Arenicola defodiens* from the sandy shores*,* and the periwinkle *Littorina littorea* from the rocky shores. This choice of species allows for representation of both shore types, as the

collection methods used, and consequently the impacts of collection can vary considerably between substrate types and target species (Fowler, 1999).

1.2 Introduction to the Literature Review

This introductory chapter aims to summarise the current state of knowledge of intertidal collection activities, the current management and legislation of these activities, and the impacts they have upon ecosystems, focussing on *Arenicola marina* (Linnaeus, 1758), *Arenicola defodiens* (Cadman & Nelson-Smith, 1993) and *Littorina littorea* (Linnaeus, 1758) as target species.

First, the background of intertidal collection, collection methods, and trends are discussed. Next, the legal framework surrounding the topic is reviewed, covering management, legislation, and description of the study site. The biology and ecology of *Arenicola marina, Arenicola defodiens* and *Littorina littorea* are then reviewed. The biology and ecology of a species must be understood if the impacts of collection are to be studied, especially when interactions at the community level could be affected. The impacts of collection on the target species and their associated communities, including bird populations, are also reviewed, identifying gaps in knowledge for each species.

1.3 Intertidal Collection

1.3.1 Food Collection - Littorina littorea

Foraging for intertidal gastropods occurs worldwide, with variety of target species (e.g. Duran and Castilla, 1989; Povey and Keough, 1991; Kyle *et al.*, 1997; Sharpe and Keough, 1998; Keough and Quinn, 2000; Roy *et al.*, 2003; Fenberg and Roy, 2012). Within the UK, the primary gastropod target species is *L. littorea* (e.g. McKay *et al.*, 1997; Cummins *et al.*, 2002; Morgan and Richardson, 2012). *L. littorea* is collected by hand as a food source, commonly commercially, and occasionally for personal use (Cummins *et al.*, 2002). They are also occasionally used for bait (Kelly, 1999) and exported live to be used as a biological anti-fouling method on oyster farms (Crossthwaite, 2012). Periwinkles are collected in large quantities, traditionally by part time fishermen and women (O'Sullivan, 1977 as cited by Cummins *et al.*, 2002).

1.3.2 Bait Collection – A. marina and A. defodiens

Bait digging is widely practiced to support both commercial and recreational fishing (Cunha *et al.*, 2005). This activity occurs globally, with a vast array of species harvested, including worms and prawns (e.g. Wynberg and Branch, 1994; Cunha *et al.*, 2005; Napier *et al.*, 2009; Sypitkowski *et al.*, 2010; Nel and Branch, 2014). The most commonly collected bait species are burrowing polychaete worms (Gambi *et al.*, 1994), including *A. marina* and *A. defodiens* in the UK. Polychaetes are often used as fresh bait by fishermen due to the fact they form part of the diet of several targeted demersal fish species (Cunha *et al.*, 2005). Both species of lugworm can be collected using a fork to dig them out of the sediment, however, only *A. defodiens* can be extracted by the use of a bait pump, which extracts with suction (Cadman and Nelson-Smith, 1993; Brind and Darbyshire, 2015). Lugworms are sometimes mechanically harvested from large bait bed areas (Beukema, 1995); however this is not carried out in the UK. In Northumberland, lugworms are often collected for bait, along with *Nereis virens* (NIFCA, 2013b). It is believed that most of the bait collection in this area is carried out by amateur anglers, however some small-scale commercial digging also occurs (NIFCA, 2013b).

1.3.3 Trends in Intertidal Collection

Global coastal collection activities are likely to increase in the near future, especially in developing countries where the human populations are expanding rapidly,

increasing the pressure on resources (Thompson *et al.*, 2002). Leisure time and disposable income have dramatically increased over the last few decades in industrialized countries, which has been associated with increased impacts from recreation activities, including intertidal collection (Thompson *et al.*, 2002).

Trends in Intertidal Collection of Food

In the developed world, subsistence gathering of food has declined over the last 50 years, linked with increasing disposable income (Fletcher and Frid, 1996; Thompson *et al.*, 2002). Despite this, Italy and the USA still have considerable collection activities occurring (Fanelli *et al.*, 1994; Murray *et al.*, 1999; Fraschetti *et al.*, 2001). The collection of food from shores is at a low intensity in the UK when compared with other countries, such as New South Wales, Australia (Underwood, 1993). The age profile of periwinkle collectors in Ireland indicates that the industry might decrease in the future, as only 18.5% of collectors were less than 40 years old, and young people perceive it as too hard work for little financial reward (Cummins *et al.*, 2002).

Trends in Intertidal Collection of Fishing Bait

The demand for wild caught bait by sea anglers in the UK is high, and it is said to be in short supply (Olive, 1999). In the 1970s it was estimated by the National Anglers Council, that 1.5 million anglers collected their own bait in the UK. Since then the number of sea anglers is believed to have decreased, possibly due to declining fishing stocks (Fowler, 1999). However, there is currently a national angling strategy in the UK, which aims to increase the participation of this sport in the future, in turn increasing the demand for bait (Angling Trust, 2013; Environment Agency, 2013). This potential increase in fishing bait demand could be added to by the changing demographics of the UK, with an influx of foreign nationals with a strong sea angling culture (Angling Trust, 2013).

1.3.4 Scale and Market Value of Intertidal Collection

Collection of *Littorina littorea*

Marine gastropods make up 2% of the molluscs fished in the world, with the UK, France and Ireland having the most important gastropod fisheries in Europe (Leiva and Castilla, 2001). One of the main species extracted from these fisheries is *L. littorea* (Leiva and Castilla, 2001), which are usually exported from the UK to the continent, where there is a large market for them, especially in France (Cummins *et*

al., 2002). The periwinkle fishery has not been well studied in England, and the market value is not well known. However, in Ireland, the periwinkle trade was estimated to be worth £5 million in 1994 (Pearson, 1994 as cited by Cummins *et al.*, 2002), with no detailed economic evaluation since. Studies in both Ireland and Scotland estimate that 4,000 tonnes of periwinkles are exported annually from each country, and around 500 part-time pickers work in Ireland (McKay *et al.*, 1997; Cummins *et al.*, 2002). However, it is difficult to accurately assess the size of periwinkle fisheries due to the unregulated, under reported, and often black market nature (Cummins *et al.*, 2002; Crossthwaite, 2012). Landings data are not a reliable estimate of collection levels in this industry, as many places which have no such data are still harvested (McKay *et al.*, 1997).

More people pick winkles when the prices are high, driven by higher demand on the continent (Cummins *et al.*, 2002). At Christmas, prices are highest £2,200 per tonne (compared to £1,400 per tonnes at other times of the year). Summer used to be the low season, however since exporting to France began, the restaurant trade still has demands in summer (Cummins *et al.*, 2002), meaning winkle picking occurs year round. Price also depends on the size of the animals, and grading (Cummins *et al.*, 2002). In 2002, pickers typically received as little as 80p per kilo, or up to £1.50 at Christmas time in Ireland, with wholesalers' prices about £2.10 per kilo for small winkles and £2.50 for larger ones (Cummins *et al.*, 2002). Currently within the BNNC EMS, wholesalers' prices average £10 per Kg (Berwick Shellfish Company, 2017; The Fish Society, 2017).

Collection of *Arenicola marina* and *Arenicola defodiens*

Total numbers of bait harvesters are difficult to ascertain due to many anglers not being associated with any formal associations/clubs (Saunders *et al.*, 1998). However, it is estimated that 2.5% of the UK population participate in sea angling annually (Watson *et al.*, 2017a), which in 2017 would equate to approximately 1.65 million individuals, of whom a significant proportion use polychaetes as bait (AFBI, 2014; Monkman *et al.*, 2015). In 1999 it was estimated that the UK used at least 1,000 tonnes of bait worms every year, with 500-700 tonnes being dug for personal use, and 300-500 tonnes by commercial bait collectors (Fowler, 1999). A lot of trade is conducted through a "black economy", meaning exact quantification of the market value is difficult (Olive, 1999). A recent assessment of the global polychaete bait

industry estimated that 121,000 tonnes are collected annually, worth £5.9 billion, with *Arenicola defodiens* listed as one of the five most expensive marine species on the global fisheries market (retail price per kg) (Watson *et al.*, 2017a). Retail values of *A. marina* and *A. defodiens* in 2017 are £4 and £53 per kg respectively (Watson *et al.*, 2017a).

1.4 Legal Framework and Management

1.4.1 The Bigger Picture

Conservation of biodiversity is globally important, and is helped with such legislation as the Convention on Biological Diversity, and the RAMSAR convention. To protect biodiversity, all countries need to act together to preserve natural ecosystems and improve biodiversity (DEFRA and England, 2013). As part of this global aim to protect biodiversity, the UK is required to have 10% of its oceans and/or coasts protected by 2020 (Boonzaier and Pauly, 2016).

1.4.2 European Legislation

As a member of the European Union (EU), the UK also has European conservation legislation to follow. The EU has specific targets for biodiversity conservation and legislation to protect key habitats and species (JNCC, 2015a). The two key EU Directives for wildlife and nature conservation are the Birds Directive (Directive 2009/147/EC on the Conservation of Wild Birds) and the Habitats Directive (Directive 92/43/EEC of 21 May 1992 on the conservation of natural habitats and of wild fauna and flora) (JNCC, 2015a). These Directives protect important species and habitats, particularly through the designation of protected sites. Under these regulations, Special Protected Areas (SPA) for birds, and Special Areas of Conservation (SAC) for habitats and other species are designated (NCAONB, 2009). Together, SPA and SAC areas form the European-wide sites known as the Natura 2000 network. An SPA and/or SAC site which incorporates a 'marine area' is called a European Marine Site (EMS), of which there are 81 in the UK (NCAONB, 2009). Within EMSs, activities need to be balanced with the ecological needs of the qualifying features (NCAONB, 2009).

1.4.3 UK Legislation

The Wildlife and Countryside Act 1981 (WCA) was designed to consolidate and amend earlier national legislation with the aim of helping to implement the Bern Convention and the Birds Directive within the UK (JNCC, 2015b). The WCA allows the designation and subsequent protection of Sites of Special Scientific Interest (SSSI). SSSIs are areas, designated by Natural England in England, which are 'of special interest by reason of any of its flora, fauna, or geological or physiographical features' (JNCC, 2015b). When a site is designated, the reasons for designation are

specified (e.g. which flora, fauna, etc. are important), and risks to these are listed. The owner or occupier of land within the SSSI must not permit or cause any of the listed risk activities, unless with permission from Natural England (JNCC, 2015b). An important amendment to the WCA, is the Countryside and Rights of Way Act 2000 (CRoW). CRoW improves measures for SSSI management, providing increased powers for site protection and threatened species (JNCC, 2010b).

The Marine and Coastal Access Act 2009 (MACAA) provides the outlines for a system for management and protection of the marine and coastal environment (JNCC, 2010c). This Act modernised inshore fisheries management, creating Inshore Fisheries and Conservation Authorities (IFCAs), with the aim of conserving marine ecosystems, whilst still enabling profitable and sustainable inshore fisheries (JNCC, 2010c). Under this legislation, IFCAs can develop and implement byelaws to protect fisheries and the marine environment (DEFRA, 2011). The Northumberland IFCA (NIFCA) is responsible for fulfilling inshore management within the study area. MACAA is additionally responsible for the designation of Marine Conservation Zones (MCZs), which protect a range of nationally important wildlife and habitats within English and Welsh territorial and UK offshore waters (JNCC, 2016).

The EU Habitats Directive is transposed into UK law by The Conservation of Habitats and Species Regulations 2010, also known as the Habitat Regulations. These regulations allow for the designation and protection of 'European sites' within the UK, with special provisions for EMSs (JNCC, 2010a). The BNNC EMS, the study site of this thesis, is designated and protected under both UK and EU law combined, and therefore appropriate management of such sites is very important to meet conservation obligations.

1.4.4 Northumberland – International, EU, and UK legislation combined

There are many different conservation designations in Northumberland, stemming from international, European, and UK legislation. There are RAMSAR sites at the international level, SACs and SPAs at the European level, and SSSIs and MCZs at the UK level. The areas often overlap and act together to protect a variety of habitats and species, forming a conservation network. The marine and intertidal conservation designations within Northumberland can be seen in Table 1:1, along with the main designated features of interest for each site (i.e. why it was designated / what is protected). The locations of each designation type can be seen in in Figure 1:1.

Table 1:1: Coastal and intertidal conservation areas within Northumberland, and their main designation/protection features, including Sites of Special Scientific Interest (SSSI), Special Areas of Conservation (SAC), Special Protection Areas (SPA), RAMSAR, and Marine Conservation Zones (MCZ) sites.

Designation Type	Site Name	Features
SSSI	Northumberland Shore	Bird aggregations – Golden plover, Purple sandpiper, Redshank, Ringed plover, Sanderling, and Turnstone
	Lindisfarne	- Bird aggregations - Little tern, Roseate tern, Bar-tailed godwit, Brent Goose, Common Socter, Curlew, Dunlin, Eider, Golden plover, Grey plover, Greylag goose, Redshank, Ringed plover, Sanderling, Shelduck, Whooper swan, and Wigeon - Supporting habitats and communities, e.g. saltmarsh, dunes, grassland, and seagrass beds - Geological designations
	Bamburgh Coast and Hills	- *Geological* – Permian Igneous rock - Habitat – Grassland of *Festuca Ovina, Agrostis Capillaris,* and *Rumex Acetosella*
	Bamburgh Dunes	- Invertebrate assemblage - Dune plant communities – 8 features
	Farne Islands	- Breeding bird aggregations – Arctic tern, Common tern, Cormorant, Eider, Guillemot, Kittiwake, Puffin, Roseate tern, Sandwhich tern, Shag - Grey seals
	Howick to Seaton Point	- Bird aggregations – Golden plover aggregations - Geological – Namurian
	Castle Point to Cullernose Point	- Breeding bird aggregations – Kittiwake - Habitat – Grassland of *Festuca Ovina, Agrostis Capillaris,* and *Rumex Acetosella,* and Reefs - Geological – Permian Igneous rock
	Alnmouth Saltmarsh and Dunes	Saltmarsh and dune communities and associated species – 12 features
	Newton Links	- Breeding bird aggregations – Little tern - Habitats – Dune and saltmarsh plant species – 11 features
	Cresswell and Newbiggin Shores	Geological designation for Westphalian and Quaternary studies
MCZ	Coquet to St. Mary's	Habitats – Low, moderate, and high energy intertidal rock, Intertidal coarse and mixed sediments, Intertidal mud, sand, and muddy sand, Intertidal underboulder communities, Moderate and high energy infralittoral rock, Moderate energy circalittoral rock, Subtidal coarse and mixed sediments

Designation Type	Site Name	Features
SAC	Berwickshire and North Northumberland Coast	- Habitats – Mudflats and sandflats not covered by seawater at low tide, Large shallow inlets and bays, Reefs, and Submerged or partially submerged sea caves - Species – Grey seals
	North Northumberland Dunes	Embryonic shifting dunes, White dunes, Grey dunes, Dunes with *Salix repens* ssp. *argentea,* Humid dune slacks, and Petalwort *(Petalophyllum ralfsii)*
SPA	Lindisfarne	- Birds – Golden plover, Whooper swan, Little tern, Greylag goose, Light-bellied brent goose, Wigeon, Ringed plover, Bartailed godwit, Redshank, Shelduck, Eider, Dunlin, Long-tailed duck, Roseate tern, Common Scoter, Red-breasted merganser, Grey plover, Sanderling. - Supporting habitats: intertidal sand and mud flats, salt marsh, seagrass beds, and rocky shores
	Northumbria Coast	Birds – Turnstone, Purple sandpiper, Little tern
	Farne Islands	Birds – Common tern, Arctic tern, Sandwhich tern
	St Abbs to Fast Castle Head	Birds – Razorbill, Herring gull, Shag, Kittiwake, and Guillemot
Ramsar	Lindisfarne	- Internationally important Birds – Waterfowl, Light-bellied brent goose, Ringed plover, Common redshank, Greylag goose, Bar-tailed godwit - Nationally important assemblages of 11 other bird species - Plants – Petalwort
	Northumbria Coast	- Internationally important bird assemblages – Little tern, Purple sandpiper, Ruddy turnstone. - Nationally important bird assemblages of 5 other species
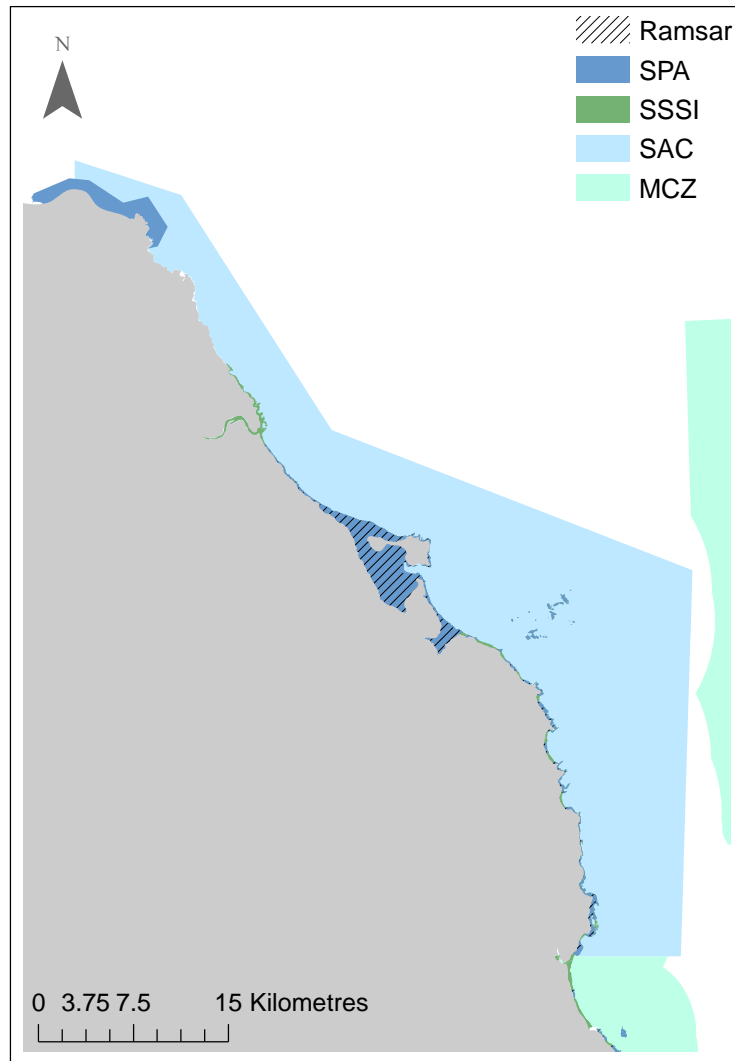
Figure 1:1: Conservation designations within Northumberland.

Many of the habitats listed in Table 1:1 are relevant to bait collection and foraging activities. Saltmarsh, sand dunes, grassland, and seagrass beds are all areas which surround or overlap with intertidal sand, mud, and rock, the source habitats of lugworms and periwinkles. As such, they are all at risk from trampling during access to collection (e.g. Hylgaard and Liddle, 1981; Andersen, 1995; Eckrich and Holmquist, 2000; Kerbiriou *et al.*, 2008; Santoro *et al.*, 2012). Within the Lindisfarne SPA, seagrass beds co-occur with lugworm beds (small patches of seagrass scattered around the sandflat – personal observation), and as such are at a significant risk of disturbance from sediment turnover directly, not only trampling. The collection habitats themselves, intertidal rock, sand, and mud (also referred to as rocky reef, under boulder communities, mudflats, sandflats, etc. in Table 1:1), and associated communities are directly relevant, with impacts from bait collection and foraging covered in detail in sections 1.6 and 1.7.

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Of the many bird species listed as designated features in Table 1:1, some are more relevant to intertidal collection activities than others. People in close proximity to birds, whatever their activity on the shore, have the potential to cause disturbance, with different species being affected in various ways and to differing degrees (Davidson and Rothwell, 1993). Impacts of bait collection and foraging on birds is discussed in more detail in sections 1.6 and 1.7 respectively. Here, potential impacts which are relevant to each designated species are summarised in Table 1:2, based on the general feeding and habitat preferences of each bird species (IUCN, 2017). Loss of prey refers to the alteration of communities from collection, not just the target species (lugworms and periwinkles) – see sections 1.6 and 1.7 for details.

Table 1:2: Potential impacts on each designated bird species within Northumberland from bait collection on sediment shores (including estuaries and mudflats), and foraging on rocky shores.

Bird Species	Bait Digging Potential Impacts			Foraging Potential Impacts		
	Breeding Disturbance	Feeding Disturbance	Loss of Prey	Breeding Disturbance	Feeding Disturbance	Loss of Prey
Little Tern	✓	✓			✓	
Purple sandpiper					✓	✓
Turnstone		✓	✓		✓	✓
Roseate Tern	✓	✓			✓	
Bar-tailed Godwit		✓	✓			
Common Scoter						
Dunlin		✓	✓			
Eider						
Golden Plover		✓	✓		✓	✓
Grey Plover		✓	✓			
Greylag Goose		✓				
Light-bellied Brent Goose		✓				
Long-tailed Duck						
Red-breasted Merganser						
Redshank		✓	✓		✓	✓
Ringed Plover		✓	✓			
Sanderling		✓	✓		✓	✓
Sanderling		✓	✓			
Whooper Swan						
Wigeon		✓				
Curlew		✓	✓			
Oystercatcher		✓	✓		✓	✓
Lapwing						
Knot		✓	✓			
Guillemot						
Cormorant						
Puffin						
Shag						
Kittiwake						
Razorbill						
Sandwhich Tern	✓	✓			✓	
Herring Gull						

1.4.5 Berwickshire and North Northumberland Coast European Marine Site

The BNNC EMS is made up of the BNNC SAC, and the intertidal area of the Lindisfarne SPA (NCAONB, 2009; NIFCA, 2013b) (Figure 1*:2*). The BNNC SAC was designated in 2000, encompassing 635 square km of shore and sea, stretching along 115km of coastline from Alnmouth up to Fast Castle Head (NCAONB, 2009). There are several interest features within the SAC (Table 1:1), however mudflats and rocky reefs, specifically intertidal rocky shores, are the most relevant to this study, being the habitats of the target species studied. Birds are also important within the BBNC EMS, being interest features of the Lindisfarne SPA (Table 1:1), which supports internationally important assemblages of rare birds and waterfowl, and high numbers of migratory species (NIFCA, 2013b).

EMS are not statutory designated areas, like SACs and SPAs; they are management units of these areas. Within the BNNC EMS boundary, there are other SPAs, which are not included under the BNNC EMS management unit, but can be seen in Table 1:1 and Figure 1:1, along with all other Northumberland marine and intertidal conservation designations.
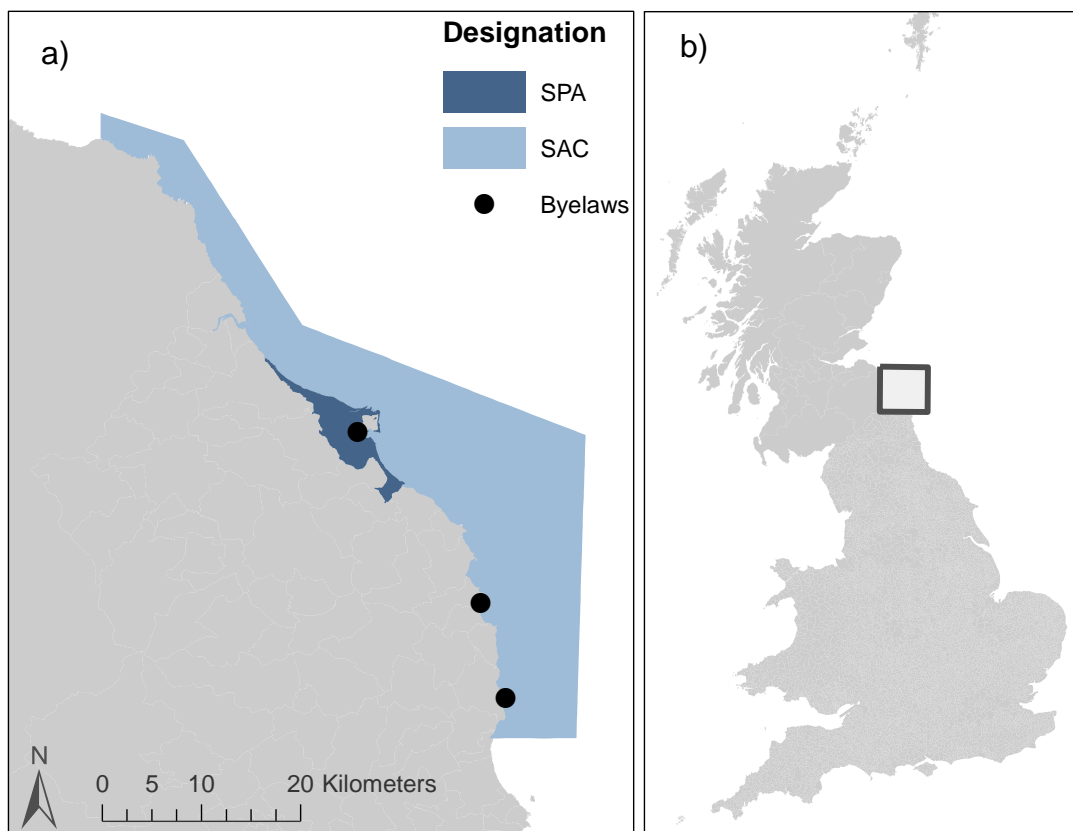
Figure 1:2: a) Special Protection Area and Special Area of Conservation which make up the BNNC EMS, and the locations of bait digging byelaws – Lindisfarne, Newton, and Boulmer from north to south. b) The location and extent of the BNNC EMS within the UK.

1.4.6 Current Regulations and Management of Intertidal Fisheries in the UK and Northumberland

Bait Collection

A. marina and *A. defodiens* collection is not regulated by fisheries legislation (Watson, 2014). There is a common law right to dig for bait as an ancillary to the right to fish, as upheld by the case of Anderson vs Alnwick DC (1992). However, this legal case further stated that the right to take bait is not unrestricted, and taking worms must be directly related to an actual or intended exercise of the public right to fish. Therefore, there is no right to take bait for commercial purposes (Watson, 2014). Personal collection of *A. marina* and *A. defodiens* can be regulated to some extent by a variety of byelaws (Watson, 2014) - competent and public bodies can exercise statutory powers to protect a habitat from potentially damaging activities within a designated site.

Bait digging within the Berwickshire and Northumberland coast EMS is currently managed with byelaws and education (NCAONB, 2009). Byelaws are present at Lindisfarne National Nature Reserve (NNR), Newton Haven and Boulmer Haven, since in the past there was significant environmental disturbance and safety problems caused by commercial bait collection in these areas (NCAONB, 2009). In the Lindisfarne NNR, Budle Bay is closed to bait digging, and has been since 1986, except for a short period in 1993 when the byelaws were challenged. Bait can be collected within a specific 'digging' zone at the Fenham Flats in the Lindisfarne NNR (UK Marine SACs Project, 2001a). At Boulmer Haven, digging has been prohibited in the area used for launching boats since 1985; however it is allowed elsewhere on the shore (UK Marine SACs Project, 2001a). At Newton Haven, a ban of bait digging was enforced by a National Trust byelaw in 1983 to protect the SSSI at the lower shore (UK Marine SACs Project, 2001a). Adherence to these byelaws has not been studied to date. It is important to study current management success before planning additional measures, to make informed decisions.

NIFCA has recently (2013) introduced a "Seagrass Protection" byelaw. Seagrasses are a sub feature of the BNNC SAC (relating to the feature of 'sandflats and mudflats not covered by water at low tide'), which are at risk from the gathering of sea fisheries

resources. The byelaw protects seagrass within the BNNC SAC to hand and mechanical gathering activities. This byelaw was introduced as part of the Defra Revised Approach to Management of Commercial Fisheries in European Marine Sites, after seagrass was identified as highly sensitive to intertidal collection activities in risk assessments, a red risk feature/fishery interaction (NIFCA, 2013a). This implies that there is scope for similar management methods to minimise other impacts associated with collection in the future, if sufficient evidence becomes available.

One education method used to influence bait digging nationally is the voluntary code of conduct created by the Angling Trust, which sets a list of rules intended to minimise the impacts of bait digging, for example back-filling holes. However, a study in the Solent showed that a code of conduct had little positive impacts on changing diggers behaviour (NIFCA, 2013a), suggesting that codes of conduct may not be a successful management tool at present.

Food Collection

Unlike the situation for marine worms, there is a public right to collect *L. littorea* both personally and commercially, as they are classified as a 'seafish' (Cummins *et al.*, 2002). As a 'seafish', commercial collection of *L. littorea* is controlled under fisheries legislation; however, currently anybody can collect them from any shore within Northumberland. There are no regulations in place to control the amount of periwinkles harvested per year, and harvesting *L. littorea* is considered a 'free for all' practice (Cummins *et al.*, 2002). The periwinkle is one of 80 non-ICES assessed stocks identified. There is inadequate information to support a harvest strategy and control rules being developed (Seafish, 2013), which has the potential to allow for over exploitation and diminished stocks without the quest for further information and details of the fishery.

However, byelaws can regulate the public right to fish. Northumberland has no periwinkle harvesting regulations or management in place currently, however, some other parts of the UK do. These are detailed in Table 1:3. Even those management measures are regarded as limited when compared to the regulation of other intertidal species such as cockles or whelks (Stranford Lough & Lecale Partnership, 2013). There is the potential for NIFCA to introduce similar periwinkle byelaws within the BNNC EMS if required.

Table 1:3: Periwinkle collection regulations in the UK (Stranford Lough & Lecale Partnership, 2013)

Region/Area	Regulation
Eastern IFCA	Minimum harvesting size of 16mm
Cornwall IFCA	Minimum harvesting size of 16mm
Devon and Severn IFCA	Minimum harvesting size of 16mm
North West IFCA	Minimum harvesting size of 16mm
Southern IFCA	Only hand gathering allowed, and closed season from 15th May to 15th September
Cumbria Sea Fisheries Committee Byelaws	Only hand gathering allowed, and 16mm minimum size
Dorset Wildlife Trust	Closed season from 15th May to 15th September

1.4.7 Potential Management of Intertidal Fisheries in Northumberland

The public right to fish is a significant issue for intertidal fisheries management, and is often considered an outdated view on modern fisheries and their environmental impacts (Boye *et al.*, 2006). There is very little formal regulation of intertidal fisheries currently (AFBI, 2013). However there are numerous possible management methods and these include voluntary guidelines and codes of conduct, byelaws for closed areas, several orders, regulating orders, licencing, weight or bag limits, size limits, and closed seasons (Underwood, 1993; UK Marine SACs Project, 2001c; Harthill et al., 2005; Boye et al., 2006; DEFRA, 2012; AFBI, 2013). The advantages and disadvantages and some key examples of each of those methods can be seen in Table 1:4.

Table 1:4: Advantages and disadvantages, and examples of each possible management method applied to intertidal fisheries

Management Method	Advantages	Disadvantages	Key Example	References
Voluntary Guidelines and Codes of Conduct	- Often secure local support - Flexible to changing conditions	- Limited success – not everyone made aware - Reliance on volunteers to police - Not great for commercial fisheries	- Bait digging in Poole Harbour and the Solent EMS - Crab Tiling in the Exe Estuary	Boye *et al.* (2006)
Byelaws / Closed Areas	- Clear basis for enforcement - Policing concentrated in small areas – easier - Easy to understand rules	- Can be difficult to enforce - Not flexible to changing conditions - Slow to establish byelaws - Displacement of activity elsewhere	- Budle Bay and Boulmer byelaws in the BNNC EMS	Boye *et al.* (2006), UK Marine SACs Project (2001b), Underwood (1993)
Several Orders	- Severs the public right to fish -Offence to remove the species listed without permission - Can set harvesting methods used	- Only for shellfish, not worms - Usually only last 10-20 years - Slow – up to 2 years to establish	- Poole Fishery Order 2015 (Southern IFCA)	(DEFRA (2012); AFBI (2013)), Boye *et al.* (2006)
Regulating Order	- Restricts fishing within an area - Allows licenses to be issued - Can set harvesting methods used	- Only for shellfish, not worms - Usually only last 20-30 years - Slow – up to 2 years to establish	- Proposed Firth of Clyde Regulating Order – prawns (*Nephrops*), and scallops	(DEFRA (2012); AFBI (2013)), Boye *et al.* (2006)
Licensing / Permits	- Monetary gain – can be used for enforcement - Can attach further conditions - Creates contact for education	- Likely cause conflict - Need high policing and education	- NWIFCA byelaw 3 – permits needed to harvest cockles or mussels. Additional minimum sizes	Boye *et al.* (2006), UK Marine SACs Project (2001c), NWIFCA (2016)
Weight or Bag Limits	- Limits the biomass removal - Acceptable for recreational collectors	- No constraint on collection effort - Difficult to enforce and educate - Difficult to set informed limit - Does not stop habitat destruction	- Eastern and North Western IFCA - 5kg of cockles and/or mussels per 24 hours	UK Marine SACs Project (2001c), Underwood (1993), Harthill *et al.* (2005)
Size Limits	-Allows all individuals to reach sexual maturity before harvesting	- Large individuals with biggest reproductive output are harvested - Policing is time consuming	- 16mm minimum periwinkle size in 5 IFCAs	Underwood (1993), Harthill *et al.* (2005)
Closed Seasons	-Prevents damage at vulnerable times, e.g. breeding	- Breeding often occurs at peak demand times, especially for lugworms	- Southern IFCA have a closed season in summer for periwinkles	UK Marine SACs Project (2001c), AFBI (2013)

1.5 Biology and Ecology of the Target Species

It is important to know the biology and ecology of the target species of fisheries to fully understand the impacts associated with their collection and their potential resilience and recoverability from harvesting. Both lugworms and periwinkles have been well studied, and their biology and ecology are generally well understood.

1.5.1 Lugworm Biology

A.marina and *A. defodiens* are closely related (Pires *et al.*, 2015) and were once considered a single species. *A. defodiens* was only described separately in 1990 (Cadman and Nelson-Smith, 1990), despite fishermen claiming their distinction from as early as 1911 (Minchin, 1911).

Distribution

A. *marina* is found throughout Europe (Watson *et al.*, 2000; Kristensen, 2001; Nielsen *et al.*, 2003; Tyler-Walters and Arnold, 2008). Within the eastern North Sea, where the largest sediment flats in the world are found, *A. marina* is one of the most dominant species (Volkenborn *et al.*, 2007a). Within the UK, *A. marina* is found on all coasts (Tyler-Walters, 2008).

Due to its relatively recent description, *A. defodiens* distribution has not been well studied (Watson *et al.*, 1998). However, it has been recorded in the western Wadden Sea, North Sea, the Skagerrak, the Westerschelde, Belgium, the Ria de Aveiro lagoon in northwest Portugal, and the North of France (Luttikhuizen and Dekker, 2010; F. Kerckhof, pers. comm., Sistermans et al., 2006, and Muller, 2004 as cited by Pires *et al.*, 2015). Within the UK they have been recorded in multiple locations in Wales (Cadman and Nelson-Smith, 1990; Cadman and Nelson-Smith, 1993), as well as several sites in Northumberland (Watson *et al.*, 1998). The actual distribution could be wider than this due to the misidentification before *A. defodiens* was described.

Description

The lugworms *A. marina* and *A. defodiens* are large, common, burrowing polychaetes (Volkenborn *et al.*, 2007a). *A. marina*, commonly referred to as blow lugworms (Watson *et al.*, 2000), usually grow to 10-25 cm in length (Riisgard & Banta, 1998 as cited in Cadman and Nelson-Smith, 1993; Kristensen, 2001).

Colouration is variable, including dark brown, red, green, black, and pink (Cadman and Nelson-Smith, 1993).

A.defodiens, commonly referred to as the black lugworm (Cadman and Nelson-Smith, 1993), are larger than *A. marina* (Cadman and Nelson-Smith, 1993). They are characteristically black, hence the common name, however they can occasionally be yellow and very rarely light brown (Cadman and Nelson-Smith, 1993; Chausson *et al.*, 2004).

Further morphological differences between the two lugworm species include: the gills (*A. defodiens* being pinnate rather than dendritic branching, longer stems, and a palmar membrane present), and the annulation pattern at the anterior end (Cadman, 1992).

Habitat

Lugworms are sedentary, inhabiting subtidal and intertidal sandy sediments (Schroer *et al.*, 2011). *A. marina* is commonly found in fine sand and muddy sand, and scarcely, or not found at all, in fine mud, gravel, and coarse sand (Callame, 1961; Bruce *et al.*, 1963; Longbottom, 1970a). *A. marina* occupy semi-permanent burrows within the upper sediment layer (Thamdrup, 1935; Flach, 1992). The burrows are 10-40cm deep and described as U-shaped (Rijken, 1979; Retraubun *et al.*, 1996a; Kristensen, 2001; Reise, 2002; Nielsen *et al.*, 2003; Volkenborn *et al.*, 2007a). The funnel at the top of the head-shaft is formed by the ingestion of sediment further down, which causes the surface sediment to sink down into the shaft (Cadée, 1976; Flach, 1992). To defecate, the lugworm moves up the burrow into the tail-shaft until the tail reaches the exit. Here the lugworm ejects its characteristic casts made up of coiled faecal strings on top of the sediment surface (Cadée, 1976). The number of casts on the sediment surface can vary with the feeding activity of the lugworms. However, it can be used as a proxy measure to record the abundance of *A. marina* (Flach and Beukema, 1994) if timed correctly.

A. defodiens burrows are deeper (up to a meter) than *A. marina*, and no feeding depressions are observed at the surface (Cadman and Nelson-Smith, 1990; Cadman 1992, as cited by Cadman and Nelson-Smith, 1993). Burrows are J-shaped, but *A.defodiens* usually lies horizontally in the burrow, rather than vertically (Fowler, 1999). The faecal cast shape is also different; *A. defodiens'* cast is smaller, neater, and spiral in shape (Cadman and Nelson-Smith, 1993). Additionally, the distribution

of *A. defodiens* within a shore is different to *A. marina*. *A. defodiens* are found further down the shore than *A. marina,* only being exposed at low spring tides (Cadman and Nelson-Smith, 1993). When *A. defodiens* and *A. marina* are found on the same shores, they each occupy distinct zones, with the black lugworm occurring lower on the shore (Cadman and Nelson-Smith, 1993). *A. defodiens* also cannot tolerate lower salinities so are absent in estuaries (Luttikhuizen and Dekker, 2010). Managers needs to consider both the differences and similarities between the two species if plans are to protect both stocks together.

Density

The densities of lugworms vary substantially, but typical densities range from 3-80 individuals per m^2 (Cadée, 1976; Jones and Jago, 1993; Volkenborn and Reise, 2006), however extremes of 150 individuals per m^2 have been observed in Northern Europe (Nielsen *et al.*, 2003). Within the UK, density has been recorded in the literature as low as 1 and as high as 38 per m^2 (Newell, 1948; Chapman and Newell, 1949; Cryer *et al.*, 1987; Olive and Cadnam, 1990). Although densities vary widely between locations, within a location the densities are relatively stable over time when compared to other infaunal species (Beukema and De Vlas, 1979; Flach and Beukema, 1994). Densities observed in lugworm populations are determined by food availability (e.g. organic matter content) and environmental factors such as sediment characteristics (Longbottom, 1970a; Flach and Beukema, 1994). Additionally, small density oscillations may occur from reproduction, predation, and migration (Reise *et al.*, 2001; Riisgard & Banta, 1998 as cited by Valdemarsen *et al.*, 2011).

Reproduction

Lugworms are gonochoristic, annual iteroparous polychaetes (Watson *et al.*, 2000). They reproduce via broadcast spawning, with quite a high dispersive potential, of around 1 to 10km (Günther, 1992; Tyler-Walters, 2008). *A. marina* is an 'epidemic spawner', which describes a local population of a single species spawning together at the same time (Watson *et al.*, 2000). The sperm is released onto the sediment surface appearing as milky white "puddles", whilst the eggs are retained in the females burrows (Duncan, 1960). The fecundity of *A. marina* is 100,000 – 1,000,000 eggs (Tyler-Walters, 2008). The eggs and young larvae develop inside the female burrow, and post larvae are capable of active migration from swimming and crawling, as well as passive movement from currents (Günther, 1992). *A. marina* reproduces from the age of 1-2

years depending on conditions and size (Newell, 1948; Duncan, 1960; De Wilde and Berghuis, 1979), and lives for approximately 5-6 years (Beukema and De Vlas, 1979), reaching maximum biomass at age 3 (Beukema, 1982).

For most British populations of *A. marina* spawning generally occurs over a few days in autumn (Duncan, 1960; Watson *et al.*, 2000). However, spawning has also been recorded in early spring (Pacey, 2000 as cited by Tyler-Walters, 2008). It is thought that lugworms need a combination of both a drop in temperature and weather conditions such as high pressure and spring tides to permit spawning (Watson *et al.*, 2000). Spawning times of *A. marina* can vary considerably, even between geographically close populations (Dillon and Howie, 1997; Watson *et al.*, 2000). *A. defodiens* reproduction is largely similar to that described for *A. marina*. However, differences include smaller oocytes, lower fecundity, and later spawning in late December to early January (Watson *et al.*, 1998; Watson *et al.*, 2008).

Feeding

A. marina is described as a sessile, head-down, subduction and conveyer-belt feeder (Kristensen, 2001; Volkenborn and Reise, 2006), and *A.defodiens* as a sand-swallowing deposit feeder (Cadman and Nelson-Smith, 1993). They are both non-selective feeders (Riisgard & Banta, 1998 as cited by Riisgård *et al.*, 1996; Papaspyrou *et al.*, 2007), assimilating ciliates, microalgae, detritus, diatoms, planktonic organisms, bacteria, and larger organisms found in the sediment and overlying water (Rijken, 1979; Andresen and Kristensen, 2002; Grossi *et al.*, 2006; Schroer *et al.*, 2011). They ingest large volumes (Longbottom, 1970a; Cadée, 1976; Kristensen, 2001; Andresen and Kristensen, 2002; Riisgard & Banta, 1998 as cited by Casado-Martinez *et al.*, 2009) of nutritionally-poor food (sediment) (Cadée, 1976; Retraubun *et al.*, 1996a; Kristensen, 2001). The consumption and following excretion of sediment displaces sediment at a rate higher than sedimentation from the water column (Cadée, 1976).

1.5.2 Lugworm Ecology

Lugworms are a major prey species for fish (Pocklington and Wells, 1992) and shorebirds (Evans *et al.*, 1979), as well as being used as bait by anglers to catch fish such as cod, whiting, haddock and flatfish (Bat, 1998; Tyler-Walters, 2008). *A. marina* and *A. defodiens* are termed habitat engineers or ecosystem engineers, meaning they alter the physical state of the habitat, and therefore affect other species (Lawton, 1994;

Wright and Jones, 2006; Volkenborn *et al.*, 2007a; Volkenborn and Reise, 2007). It is well known that biogenic habitat transformations structure benthic assemblages, possibly extending over wide spatial scales (Reise, 2002; Volkenborn and Reise, 2007). *A. marina* inflicts substantial impact on the sediment by reworking it (Retraubun *et al.*, 1996a). It is estimated that *A. marina* mixes the upper 6-33cm of the sediment in the Dutch Wadden Sea per year, which is similar to many other estimates from different areas (1-18cm per year) (Cadée, 1976; Retraubun *et al.*, 1996b; Risgard & Banta, 1998 as cited by Valdemarsen *et al.*, 2011). This substantial reworking destabilises the sediment, which has negative effects on some macrobenthic species abundance, primarily sedentary species (Woodin, 1985; Brey, 1991; Flach, 1992; Volkenborn *et al.*, 2007a; Volkenborn and Reise, 2007).

However, lugworms also have positive effects on other organisms. The burrows form biogenic structures, doubling the sediment-water interface area, transporting particles, dissolved metabolites and oxygen through the sediment like veins (Reise, 2002). Due to the ventilation, the inside of the burrow wall is oxidised compared to the surrounding sediment, forming a unique microhabitat (Banta *et al.*, 1999; Nielsen *et al.*, 2003). This process also extends the Redox Potential Discontinuity layer deep into the otherwise anoxic sediment, aerating the environment for other subsurface species (Baumfalk, 1979; Retraubun *et al.*, 1996a; Schroer *et al.*, 2011). Many species live inside lugworm burrows, as well as in the casts and funnels at the surface, forming different species assemblages to those in the surrounding sediment (e.g., Reise, 1981; Reise, 1987; Brey, 1991; Flach and Beukema, 1994; Reise, 2002). Therefore, lugworms play a key role in developing the benthic community structure (Brey, 1991).

Lugworm sediment reworking and burrows have both negative and positive impacts upon other sediment dwelling species, some examples of which can be seen in Table 1:5. However, it is believed that lugworms may have a diversifying effect on the marine benthos overall (Reise, 2002). This important role must be considered in management plans for intertidal collection, as overexploitation of lugworms has the potential to not only impact upon the target species populations, but also the overall biodiversity of sediment shores and mudflats.

Table 1:5: Organisms impacted by lugworm burrows and reworking: positive (e.g. increased abundance or distribution, habitat creation, increased oxygen levels, etc.) or negative (destabilised sediment, less food availability, etc.).

	Organisms	References
Positively impacted	Multiple species of flatworm, *Bathyporeia sp,* nematodes	(Reise (1987); Brey (1991); Wetzel *et al.* (1995))
Negatively impacted	*Corophium volutator*, Eelgrass, juvenile *Scoloplos armiger, Pygospio elegans, Capitella capitate, Cerastoderma edule, Macoma balthica*	(Flach (1992); Philippart (1994); Valdemarsen *et al.* (2011))

1.5.3 Periwinkle Biology

Distribution

Littorinids are common throughout the world (Geller, 1991; Mill and Mcquaid, 1995). *L. littorea* are common inhabitants of the intertidal zone of the North Atlantic (Perez *et al.*, 2009; Storey *et al.*, 2013), and are found frequently on coasts of Western Europe and Northeast America (Barnes and Hughes, 2009). The overall distribution ranges from Northern Spain to the White Sea in Europe (Fretter & Graham, 1980, Bequaert, 1943 as cited in Johannesson, 1988; Jackson, 2008b), and New Jersey to Labrador in Northern America (Fretter & Graham, 1980 as cited in Johannesson, 1988). In Britain, *L. littorea* are found on all coasts, apart from the Channel Isles and Isles of Scilly (Jackson, 2008b).

Description

The marine gastropod *Littorina littorea* (Linnaeus, 1758)*,* also known as the common or edible periwinkle, is the largest British Periwinkle species, reaching a maximum shell height of 52mm (Jackson, 2008b). They are coiled mesogastropods (Geller, 1991), occurring in a range of colours, but are usually grey-brown or black, with lighter shades towards the apex (Jackson, 2008b). They are one of the most abundant gastropods on UK shores (Moore, 1937).

Habitat

A variety of habitats are inhabited by *L. littorina* within the intertidal zone - rocks, stones, gravel, soft mud, and sand (Moore, 1937; Jackson, 2008b). However, it is usually absent or rare on unstable substrate such as shingle and unconsolidated sand (Evans, 1947). *L. littorea* is one of the only littorinid species which is found commonly on both soft bottom and hard substrate environments (Bandel, 1974). Despite this

variety, they are most abundantly found on rocky and stone shores (Smith and Newell, 1955; Storey *et al.*, 2013), within which they are widely distributed in all but extremely exposed areas (Jackson, 2008b). *L. Littorina* has a characteristic intertidal distribution, being more abundant in the littoral zone, and scarce in the subtidal (Perez *et al.*, 2009). Within the intertidal zone, *L. littorea* occur at all shore levels, however preferentially at low shore levels (Norton *et al.*, 1990; Perez *et al.*, 2009)

Density

L. littorea can reach densities of several hundred (Janke, 1990; Wilhelmsen and Reise, 1994; Carlson *et al.*, 2006), or even thousands (Vadas, 1992; Buschbaum, 2000; Eschweiler *et al.*, 2008) of individuals per square meter. In the UK, densities are usually below 200 per square meter (Norton *et al.*, 1990), with higher densities generally found in North America (Petraitis, 1987) due to lower levels of competition (Brenchley and Carlton, 1983). Within a shore type, biological and environmental influences affect the abundance, distribution and size of periwinkles on a shore. *L. littorea* exhibit highly variable zonation patterns, displaying complex size gradients (Smith and Newell, 1955; Williams, 1964; Vermeij, 1972), with larger individuals often found further down the shore and into the subtidal (Perez *et al.*, 2009). On rocky shores, *L. littorea* distribution positively correlates with the bare rock percentage cover and rugosity (Carlson *et al.*, 2006). More rugose sites are thought to be favoured because of the refuge they provide from predators and the availability of damp, shaded areas to minimize desiccation and thermal stress when exposed at low tide (Carlson *et al.*, 2006). *L. littorea* tend to form clusters, aggregating in areas with more favourable conditions such as rock pools (Newell, 1958).

Reproduction

L. littorea live for 5-10 years, with an age of maturity of around 2-3 (Jackson, 2008b), or 3-4 years (Fish, 1972) depending on conditions such as food availability . The size at sexual maturity differs geographically, with shell heights ranging from 12mm observed in Wales (Williams, 1964; Fish, 1972) to 17mm in Plymouth (Moore, 1937). Fecundity of *L. littorea* increases with size, and therefore age (Hughes and Answer, 1982). However, parasite infection reduces fecundity in *L. littorea,* by converting them from iteroparous to semelparous organisms (Hughes and Answer, 1982). Infection incidence increases with *L. littorea* age (Hughes and Answer, 1982), and therefore in

heavily parasitized populations, young snails may produce the most eggs (Robson and Williams, 1971).

L. littorea are annual episodic spawners (Jackson, 2008b), and the majority of spawning occurs in March and April (Grahame, 1975). However, there are geographical differences in spawning months (Fish, 1972). *L. littorea* are planktonic spawners, with a dispersal potential of more than 10km (Jackson, 2008b), with a maximum distance of 300km predicted (Johannesson, 1988). Therefore recruitment may not be from the local population, and considerable gene flow can occur between separated populations (Berger, 1973). However, gene flow between populations can be reduced in certain conditions, such as areas with dense vegetation trapping eggs (Fish, 1972). This means that some populations under certain conditions may not recruit from any other populations, only their own. These populations may be more susceptible to over exploitation, and as such may require more management.

Feeding

The common periwinkle is a generalist intertidal herbivore (Imrie *et al.*, 1989), using a taenioglossan radula to feed on a variety of food items, from macroalgae including filamentous and foliose algae, to microalgae including non-siliceous microalgae and diatoms (Steneck and Watling, 1982; Sommer, 1999b). *L. littorea* preferentially consume early successional and ephemeral algae, such as *Ulva lactuca* (Lubchenco, 1983; Watson and Norton, 1985; Barker and Chapman, 1990; Norton *et al.*, 1990). They are thought to generally avoid mature leathery macrophytes such as *Fucus* sps, even under conditions of nutritional duress (Steneck and Watling, 1982; Watson and Norton, 1985).

1.5.4 Periwinkle Ecology

Herbivores play a key role in marine ecosystems, affecting the composition, diversity and biomass of primary producers, with large potential impacts on ecosystem functioning (Griffin *et al.*, 2010). On rocky shores, grazers are fundamental in controlling abundance and distribution of algae (Lubchenco and Gaines, 1981; Hawkins and Hartnoll, 1983; Vadas, 1992; Anderson and Underwood, 1997). In particular, periwinkles influence benthic community structure and the recruitment success of seaweeds that are structurally important in the habitat (Janke, 1990; Wilhelmsen and Reise, 1994; Buschbaum, 2000).

It is widely accepted that disturbance at intermediate levels can enhance diversity as dominant competitors are removed, and competition relaxed by reducing biomass (Paine and Vadas, 1969; Osman, 1977; Connell, 1978; Aronson and Precht, 1995; Sommer, 1999a; Shea *et al.*, 2004). When grazing is considered a disturbance, primary producer diversity can be maximised with intermediate grazing activity, as observed for *L. littorea* grazing by Sommer (1999a). In the absence of grazing, competitive exclusion is thought to diminish diversity, making *L. littorea* presence a key factor contributing to biodiversity preservation on rocky shores. *L. littorea grazing* may also help to control the dominance of opportunistic macroalgae from increased nutrient levels (Diaz *et al.*, 2012), and invasive species such as *Codium fragile,* via the grazing of new recruits (Scheibling *et al.*, 2008). This could protect coastal habitats by buffering eutrophication effects and controlling the spread of invasive algal species by exerting top-down control (Scheibling *et al.*, 2008; Diaz *et al.*, 2012).

L. littorea grazing can also impact upon other organisms in the community. The removal of algae by grazing has been seen to cause both direct and indirect impacts on sessile organisms (Petraitis, 1983; Bertness, 1984; Petraitis, 1987; Vadas, 1992; Anderson and Underwood, 1997; Buschbaum, 2000). Also, direct damage of sessile organisms can be caused by the physical disturbance that grazing inflicts (Dayton, 1971; Denley and Underwood, 1979; Hawkins and Hartnoll, 1983; Petraitis, 1983; Underwood *et al.*, 1983; Farrell, 1988; Buschbaum, 2000). Examples of both algae and sessile organisms impacted by periwinkle grazing can be seen in Table 1:6.

Body size of grazers influences the affect they have on the community: grazing rates and habitat selection are size-dependant (Geller, 1991; Saier, 2000). Consequently, factors which influence size distribution of periwinkles are important for community dynamics (Eschweiler *et al.*, 2009). It is important to consider the wider ecosystem impacts of periwinkles and their grazing activity in intertidal collection management plans, as overexploitation and depletion of target species stocks has the potential to indirectly influence biodiversity.

Table 1:6: Organisms impacted by periwinkle grazing: positive (e.g. reduced competition, more open space, etc.) or negative (bulldozing effects, reduced settlement, competition for food, etc.).

	Organisms	References
Positively impacted	Mature fucoids, fucoid dependent communities	Lubchenco (1983)
Negatively impacted	Ephemeral algae, *Codium fragile*, barnacles	(Petraitis (1987); Janke (1990); Albrecht (1998); Buschbaum (2000); Scheibling *et al.* (2008))

1.5.4 Implications for management

Biology

The population biology of both lugworms and periwinkles make them relatively resilient to harvesting and place them low on the conservation radar. Both are very common and widely distributed throughout the UK and Europe (Geller, 1991; Nielsen *et al.*, 2003; Storey *et al.*, 2013). Their conservation and management appears less critical than for rarer or endangered species, due to conservation priorities often focussing on measures of 'irreplaceability' (Brooks *et al.*, 2006). This may explain why regulations and management have largely overlooked both fisheries to date. Both lugworms and periwinkles produce a high number of offspring, have relatively short life cycles, mature quickly, and have high offspring dispersal potential (Johannesson, 1988; Günther, 1992; Watson *et al.*, 2000; Jackson, 2008b); all the population parameters of an *r*-selected species (Adams, 1980). This implies that they may be fairly resilient to harvesting and disturbance compared to K-selected fishery species, which are highly sensitive to overfishing and recover more slowly (Adams, 1980). Despite these positive biological parameters, over exploitation of lugworms and periwinkles at a local scale has the potential to harm or threaten stocks (Shahid, 1982; Beukema, 1995; Berthelon *et al.*, 2004), and they should be considered in management plans alongside other fisheries.

Disturbance could have the largest impact upon a population's reproduction during times of spawning. For this reason, fishery closures during spawning seasons are commonplace (van Overzee and Rijnsdorp, 2014). Since the exact spawning times of individual populations vary for all three species, it is important that management aimed at protecting spawning populations (e.g. closed seasons) is either broad enough to cover all possibilities, or individual populations are studied in depth to gain accurate spawning dates for effective protection. Similarly, accurate localised age of

maturity data needs to be established for both fisheries within Northumberland, for management methods such as minimum harvest sizes to be established most effectively (McIntyre *et al.*, 2015).

Ecology

The impacts of periwinkles as grazers, and lugworms as bioengineers are significant in the rocky and sediment shore habitats and communities (e.g. Lubchenco, 1983; Janke, 1990; Flach, 1992; Volkenborn *et al.*, 2007a; Griffin *et al.*, 2010). These effects are ephemeral and require renewal to be effective (Reise, 2002), meaning that if over exploited, the habitats and communities would undergo changes as well as the target stock. Similarly, predators could experience the effects of fishing, such as birds and fish (generally better recognised in termed of conservation importance) which rely on healthy populations of lugworm and periwinkles as important prey species (Evans *et al.*, 1979; Pocklington and Wells, 1992; Masero *et al.*, 2008). Management plans must consider the indirect impacts of intertidal harvesting also, and those occurring locally within the BNNC EMS require assessment.

1.6 Impacts of Lugworm Collection

The majority of reports suggest that the collection of bait has significant detrimental effects on wildlife (Berthelon *et al.*, 2004).

1.6.1 Impacts of bait collection on Lugworms

Bait digging can act as a form of selective predation, as diggers preferentially take *A. marina* and *A. defodiens* over other worm species, and remove the largest individuals (Shahid, 1982). Some collectors do not limit the size or number of worms they take, and sometimes exploit nursery grounds (Fowler, 1999). Bait digging causes mortality in lugworm populations, which can lead to reduced abundance and stock declines (Blake, 1979a; Beukema, 1995). Additionally, the size structure of lugworm populations can be altered (Shahid, 1982). Some details of the impacts observed in previous studies are summarised in Table 1*:7*. The severity of impacts upon the target species appears to be correlated to the digging intensity.

Lugworms are generally considered to be fairly resilient to bait collection activities, with life strategies allowing for high recoverability (Olive, 1993; Spikes, 1993). Despite this, lugworm recovery rates vary considerably between studies (as seen in Table 1*:7*), influenced by collection intensity and environmental differences between shores. For example, stocks may be more seriously impacted by digging if they are isolated (e.g. small pocket beaches), as recruitment and migration of nearby stocks may not be possible (Fowler, 1999). *A. defodiens* may be more resilient to bait collection than *A. marina* because of its' subtidally extending distribution. This subtidal part of the population is not accessible to harvesters (Fowler, 1999), and therefore could act as a refuge, allowing continued recruitment, and the migration of adult worms into disturbed areas (Rees and Eleftheriou, 1989; Spikes, 1993).

Overall, lugworm abundance and size structure can be impacted by bait collection, however, recoverability can be high and fast under the right circumstances, where the collection intensity is low, the population is not isolated, and refuge stocks are available. Van den Heiligenberg (1987) suggests that the extinction of lugworm stocks seems impossible if the digging activity is localised (i.e. on just one shore or area of shore).

Table 1:7: The impacts and recovery observed for Arenicola marina and Arenicola defodiens in previous lugworm exploitation studies

Intensity of Collection	Method	Impacts	Recovery	Reference
High	Mechanical	Doubled mortality rate and stock decline	Slow – 3 years to reach original ratios	Beukema (1995)
High	Digging	Population crash – reduced abundance	Medium – Increased abundance after a 2 year ban	Olive (1993)
High	Digging	Reduced average and max size	N/A	Shahid (1982)
Medium	Digging	Reduced lugworm abundance inside dug areas	Medium/Slow – No repopulation within 6 months study (whilst continued monthly disturbance	Cryer *et al.* (1987)
Low-Medium	Digging	No significant reduction in lugworm abundance and spawning population not impacted	Fast – Within 1 month	Blake (1979a)

Table 1:8: The impacts and recovery observed for sediment communities in previous bait collection studies. Direct = disturbance, Indirect = consequences of reduced lugworms

Direct or Indirect Impact	Taxa	Impacts Described	Reference
Direct	*Littorina littorea*	Negative - Burial under sediment piles with high fatality	Chandrasekara and Frid (1998)
	Hydrobia ulvae	Negative - Burial under sediment piles with medium fatality	Chandrasekara and Frid (1998)
	Cerastoderma edule	Negative - Population crash from deep burial	Jackson and James (1979)
	Sabella worms	Negative - Beds uprooted by digging	Dyrynda (1995)
	Zostera sp	Negative - Beds uprooted by digging	Dyrynda (1995), Mieszkowska (2010)
	Small surface dwelling polychaetes	Negative – reduced abundance	Brown and Wilson (1997)
	Euphausia brevis	Negative – increased heavy metal content inside them	Howell (1985)
	Mya arenaria	Negative – almost locally extinct	Beukema (1995)
Indirect	Predacious and tube-building worms	Positive – increased abundance	Volkenborn and Reise (2007)
	Sub-surface deposit feeders	Negative – reduced abundance	Volkenborn and Reise (2007)
	Juvenile *Scoloplos armiger*	Negative – loss of lugworm tail shafts where they gather	Volkenborn and Reise (2007)
	Nereis diversicolor	Positive – increased abundance from more stable and nutritious sediment	Volkenborn and Reise (2006)

1.6.2 Impacts of bait collection on non-target species

Non-target species are often affected by the activities of bait diggers, impacting sediment communities on an ecosystem level. In 2006, a Defra report stated that bait digging was a high threat to marine biodiversity (AFBI, 2009). The physical disturbance of the sediments is a direct impact, and the removal of lugworms and their ecosystem engineering effects is an indirect impact.

Direct Impacts – Sediment Turnover and Trampling

Physical disturbance from sediment turnover can directly damage and kill infauna, or bury them within the sediment to depths were they may be incapable of surviving (Chandrasekara and Frid, 1998). Additionally, sediment turnover can disrupt the sediment layers, releasing pollutants from the anoxic layer, and increasing the heavy metal content (Howell, 1985). Digging can also reduce the amount of organic matter within the sediment (Watson *et al.*, 2017b), diminishing food availability for many species. The total biomass of infauna has been seen to reduce substantially after digging events; up to a 40% reduction from hand digging (Van den Heiligenberg, 1987; Brown and Wilson, 1997), with impacts being cumulative over time, and even observed at low digging intensities (Brown and Wilson, 1997). Lower variability in macrofaunal species compositional structure, and increased β diversity (variation) have also been observed in dug areas (Watson *et al.*, 2017b). Some species are more sensitive to bait digging disturbance than others. For example, species with limited burrowing (Chandrasekara and Frid, 1998), and delicate species (Beukema, 1995). Diverse examples of taxa observed to be directly impacted by bait worm collection disturbance can be seen in Table 1:8.

The method of collection can alter the severity of direct impacts. Using a bait pump to harvest *A. defodiens* rather than the traditional digging method creates less sediment disturbance and turnover. The disturbance is concentrated in a small column of sediment directly around the lugworm (Fowler, 1999) rather than over a large area, meaning that a smaller volume of sediment is disturbed, and no spoil heaps are produced to bury organisms. Therefore bait pumps appear to cause minimal direct impacts compared to digging. Mechanical harvesting of lugworms is the most destructive method, causing substantially more deaths of benthic fauna per lugworm harvested, than hand gathering methods (Van den Heiligenberg, 1987). Within hand digging, the smaller details of the method are also important to consider. Some

experienced hand diggers make trenches which they back-fill as they go, whilst less-informed bait diggers tend to dig scattered holes which are left open (Fowler, 1999). Back-filling of dug areas minimises the disturbance and therefore has a lower impact (Kaiser *et al.*, 2001). Additionally, the intensity of collection can also influence the severity of impacts, with the impact being proportional to the intensity of digging, i.e. commercial gathering resulting in larger impacts than casual gathering of bait (Anon, 1992 as cited by JNCC and Natural England, 2011).

Trampling of sediment shores by bait diggers can also directly impact upon the sediment community, even in areas that are not dug. Trampling can kill and bury infauna, as well as altering sediment properties (Rossi *et al.*, 2007). However, since trampling is inflicted on the sediment communities by many different shore users, not just bait collectors, the impacts are not explored further here, or in this thesis as a whole.

Indirect Impacts – Reduced Lugworm Abundance

If lugworm stocks were to be reduced or extinct locally by overexploitation, there would be knock-on changes to the ecosystem as a whole. These indirect impacts of bait removal can be considerable and far-reaching (Cryer *et al.*, 1987). The sediment community structure has been seen to alter, with various species reacting differently to the altered sediment characteristics without the presence of lugworms and their bioengineering. Some species abundances increase, whilst others decrease in response to the removal of lugworms from a shore (Volkenborn and Reise, 2007). Some examples of these indirect impacts upon different taxa can be seen in

Table 1:*8*. The habitat alterations causing the community shifts include the accumulation of: microphytobenthic biomass, inorganic nutrients, organic matter and fine particles, and ammonium, silicate, sulphide, and phosphate concentrations in the pore water when lugworms are no longer present (Volkenborn *et al.*, 2007a; Volkenborn and Reise, 2007).

Recovery of the sediment community usually occurs within 1 year, from recruitment of juveniles into disturbed areas. However it can be a lot faster (Van den Heiligenberg, 1987), or a lot slower (Beukema, 1995), depending on the method and the intensity. Slower growing species such as large bivalves and burrowing echinoderms are the slowest to recover (Beukema, 1995), whilst other species can recolonize relatively quickly (Van den Heiligenberg, 1987).

Impacts on Birds

Birds are also susceptible to the effects of bait digging (Masero *et al.*, 2008). Bird disturbance is considered one of the most serious impacts of bait collection in British estuaries over winter (see Davidson and Rothwell, 1993 for a detailed review). Unfortunately peak bait worm demand in winter coincides with the presence of over-wintering and migrating populations of wildfowl and waders with international importance (Townshend and O'Connor, 1993). Migratory birds are particularly vulnerable to disturbance due to their reliance on a few coastal areas during their journey (Skagen and Knopf, 1993; Masero *et al.*, 2008). The presence of bait diggers can drive off roosting or feeding birds from the shore (Evans and Clark, 1993; Watson *et al.*, 2017b), and the most frequently used shores can be almost permanently unsuitable for birds, as diggers can disturb the feeding activities of birds over many thousands of meters (Van den Heiligenberg, 1987). The disturbance of birds can lead to them searching for new feeding areas, increasing energy expenditure and food competition, and ultimately leading to increased winter mortality rates in some cases (West *et al.*, 2002; Masero *et al.*, 2008). The habitat loss impacts for birds can last longer than the time the diggers are present, as trenches left behind can remain flooded and unsuitable for foraging activities (Fowler, 1999).

As well as impacting birds through disturbance, bait diggers can also reduce the abundance of the birds prey species, through the reduction in invertebrate biomass or size after sediment turnover / digging (Van den Heiligenberg, 1987; Bowgen *et al.*, 2015). How much birds are impacted by these alterations in food supply will depend upon the ability to switch prey and/or foraging area, as well as environmental factors and the intensity of harvesting (Masero *et al.*, 2008). Alternatively, a positive impact from sediment turnover is that it can bring infauna to the surface where it is more vulnerable to predators including birds, making prey species more accessible after the bait digging has ceased. For example, oystercatchers have been observed to be attracted to recently dug areas, where they can eat cockles off the surface (Jackson and James, 1979).

Within Northumberland, birds have been considerably impacted by bait digging in the past. When Budle Bay (within the Lindisfarne SPA) was closed to bait collection as a trial, bird numbers increased, and later decreased significantly when the ban was reversed (Fowler, 1999). Subsequently, after the lasting ban in 1986, bird populations

have recovered considerably. Within the Lindisfarne SPA, the presence of bait diggers has been recorded to greatly reduce shore use by waterfowl, particularly wigeon (Townshend and O'Connor, 1993).

1.6.3 Summary and Implications for Management

Lugworms, the sediment community as a whole, and birds can be negatively impacted by bait worm collection, with impact intensity and subsequent recovery rates dependent upon various factors such as: extraction method, collection intensity, and shore condition and features (e.g. Van den Heiligenberg, 1987; Rees and Eleftheriou, 1989; Beukema, 1995).

The habitats subject to bait digging, sand flats and mud flats, are a feature of the BNNC SAC, and as such, activities which negatively impact upon the interest feature need to be managed (MMO, 2014b). This is especially true for birds, being the interest features for the Lindisfarne SPA, and other SPAs falling within the BNNC EMS boundary (NCAONB, 2009). Since the sediment ecosystem as a whole has been observed to be impacted by bait collection elsewhere (Volkenborn and Reise, 2006; Volkenborn *et al.*, 2007a; Volkenborn and Reise, 2007), it is important to research these impacts locally, and introduce management measures where appropriate.

Within sediment shores, invertebrates appear to be most at risk from bait digging impacts, experiencing high death rates, and for certain species, slow recovery (Van den Heiligenberg, 1987; Beukema, 1995). In terms of management, this creates the difficulty of the major impacts being invisible to collectors. Conservation success is often founded on local support, which is strongly influenced by perceptions of the impacts (Bennett and Dearden, 2014). If collectors cannot directly observe and appreciate the impacts claimed by scientists and managers, local support and compliance of management measures may suffer as a result.

The contrasting lugworm harvesting methods should be considered in management plans. Bait digging is more destructive than bait pumping (Fowler, 1999), and so management measures could direct more collectors towards bait pumps. However, this would be difficult since *A. defodiens* is not as common as *A. marina* (Cadman and Nelson-Smith, 1993). Increased harvesting focus on *A. defodiens* could lead to overexploitation of the target species. Alternatively, more focus on promoting back-filling with a code of conduct and education could help to reduce impacts of digging

and increase recovery rates, although the effectiveness of this management method is questionable (Watson *et al.*, 2015). Distinction between commercial and recreational lugworm collection would be useful for management, but remains a challenge and topic of debate within the industry (Watson *et al.*, 2017a). Since only personal collection is allowed by law, being able to distinguish and subsequently control commercial collection would reduce the intensity of lugworm collection, with the aim of reducing the severity of impacts (Anon, 1992 as cited by JNCC and Natural England, 2011).

Management plans also have the potential to focus protection on particularly sensitive species. For example, larger, slow recovering species, such as burrowing heart-urchins and bivalves (Jackson and James, 1979; Beukema, 1995), could be protected from digging disturbance by restricting bait collection to areas where these species primarily do not occur. This method of management has already been used for seagrass within the BNNC EMS, with areas covered by seagrass closed to the exploitation of fisheries resources, including bait digging (NIFCA, 2013a), suggesting that it is achievable for other sensitive species locally.

In summary, there is considerable potential for management to reduce the impacts observed from lugworm collection, however, further study is required to fully inform management plans. Currently, knowledge gaps remain within the BNNC EMS, with data lacking on the direct and indirect impacts occurring at present local harvesting levels. Research is needed with regard to the target species as well as the associated sediment communities.

1.7   Impacts of Periwinkle Collection

*1.7.1   **Impacts of collection on Periwinkles***

Exploitation of rocky shore organisms can have major impacts on the target species, with declining populations observed in previous studies (Thompson *et al.*, 2002). Harvesting can affect the density of target species; by over-collection, removal of the most fecund individuals, or habitat damage (Berthelon *et al.*, 2004). However, many studies have failed to observe reduced periwinkle abundance related to present exploitation levels (see Table 1:9 for examples). When density effects are not observed, this is not evidence that there is no effect upon that population, as reductions in the reproductive fitness of one population may be masked by the larval supply of other healthy populations (Berthelon *et al.*, 2004).

Collection of periwinkles can be regarded as selective predation (Sharpe and Keough, 1998), being capable of causing a shift in the modal size of populations (Berthelon *et al.*, 2004). When recruitment is high, and large individuals are removed preferentially, the average size of individuals can decrease leading to growth overfishing (Thompson *et al.*, 2002). This has been observed for both periwinkles and similar rocky shore molluscs (Table 1:9). However, size impacts can be masked by collectors choosing to harvest from shores with the largest periwinkles present, which could explain why Berthelon *et al.* (2004) observed larger periwinkles on collected shores (Table 1:9). A reduction in periwinkle size may have less severe effects on the reproductive fitness of a population than in other species, due to the highest reproductive output coming from smaller individuals in parasitized populations (McKay *et al.*, 1997).

Periwinkles are abundant and mobile, so populations are unlikely to be significantly impacted by short-term, localised collection (Crossthwaite, 2012). Large differences in the intensity of impacts observed between studies (Table 1:9) suggests that local differences between shores, including the intensity of collection, influences the impacts harvesting of periwinkles has on the target species. The resilience of a population is influenced by the availability of refuge populations, age of maturity, and the size of a population (Berthelon *et al.*, 2004).

Table 1:9: The impacts and recovery observed for the target species in previous periwinkle or mollusc exploitation studies

Impact Category	Impacts Described	Reference
Abundance	Current abundances impacted by historical/background collection	Quigley (1999), Crossthwaite (2012)
	No impacts on abundance between collected and non-collected sites	Berthelon *et al.* (2004)
	No impacts on abundance with 12 weeks simulated collection	Crossthwaite (2012)
Size	Smaller mean size on collected shores	Quigley (1999)
	Highest intensity harvested shores contained the smallest winkles and bimodal body size distribution	Crossthwaite (2012)
	Most rocky shore mollusc target species had larger sizes inside protected areas	Keough *et al.* (1993b)
	No impacts observed on mollusc size between collected and non-collected shores	Keough and King (1991)
	Largest periwinkles at harvested shores, but fewer juvenile periwinkles present	Berthelon *et al.* (2004)

Table 1:10: The impacts and recovery observed for rocky shore communities in previous intertidal harvesting studies. Direct = disturbance, Indirect = consequences of reduced periwinkles

Direct or Indirect	Taxa	Impacts Described	Reference
Direct	Community	No impact – community remained the same after 12 weeks	Crossthwaite (2012)
		Negative – altered community structure from long term background harvesting – lower species richness	
	Community	Negative – different community structure at collected shores	Quigley (1999)
	Semibalanus balanoides	Negative - Lower abundances on exploited shores	Berthelon *et al.* (2004)
	Fucoids and Foliose algae	Negative – reduced abundance from trampling	Berthelon *et al.* (2004)
	Mussels and barnacles	Negative – dislodged or crushed by trampling	Fowler (1999)
Indirect	*Acophyllum nodosum*	Positive – lower periwinkle densities increased germling survival	Cervin and Aberg (1997), Cervin *et al.* (2004)
	Notoacmea testudinalis	Positive – increased growth and survival when periwinkles excluded	Petraitis (1989)
	Algae	Positive – increased cover	AFBI (2009), Buschbaum (2000)
		No impact – no change due to increase in other grazers	Lindberg *et al.* (1998)
	Semibalanus balanoides	Positive – increased survival of recruits	
		Negative – reduced growth rates due to increase in algae	Buschbaum (2000)
	Ascophyllum nodosum and *Porcellana platycheles*	Positive – most abundant on shores with low periwinkle abundance	Crossthwaite (2012)
	Patella vulgata	Positive – increased abundance from less competition	Quigley (1999)

1.7.2 *Impacts of collection on non-target species*

Rocky shore harvesting impacts can be significant, being highly species dependent (Crowe *et al.*, 2000). On harvested rocky shores, due to selective predation, communities tend to converge towards a common state of abundance and diversity (Philip and Bosman, 1986), reducing the diversity between regions (Sharpe and Keough, 1998). There are both direct and indirect impacts on non-target species from periwinkle collection.

Direct Impacts – Boulder Turning and Trampling

Direct impacts of periwinkle collection on the rocky shore community are due to the disturbance created when harvesting occurs, causing physical damage to both plants and animals (Berthelon *et al.*, 2004). This includes boulder turning and trampling. Some harvesters turn the rocks over to look for periwinkles, often leaving them upturned (AFBI, 2009). Boulder or stone turning damages the diverse under-boulder communities which require stable boulder habitats, relying on the shelter the boulders provide, whilst other organisms depend on the upper rock surfaces, such as seaweeds (Liddiard *et al.*, 1989). Turning boulders reduces habitat stability, can directly crush and kill fauna, smother algae, and leaves under-boulder communities exposed to desiccation, predation, and wave action when left upturned (Berthelon *et al.*, 2004). If rocks are turned frequently, and not returned to their original positions, the habitat stability and biodiversity can be reduced (Davenport and Davenport, 2006).

Trampling over rocky shores to collect intertidal species has been shown to affect species composition due to the physical contact and wear it creates (Tyler-Walters and Arnold, 2008). Trampling can reduce biodiversity, abundance, and biomass (JNCC and Natural England, 2011), creating paths with low algal cover and a higher percentage of bare rock (Berthelon *et al.*, 2004; Tyler-Walters and Arnold, 2008). The effects of trampling can be seen even at low trampling intensities and impacts can persist for several years (Povey and Keough, 1991). However, results of trampling can be very variable, and the impacts appear to depend on the intensity, duration, and frequency of the trampling, as well as the nature of the receiving habitat, and even the type of footwear used (Tyler-Walters and Arnold, 2008; JNCC and Natural England, 2011).

These disturbances by intertidal harvesters can negatively alter the abundances of species and overall community structure present on a shore, key examples of which can be seen in Table 1:10. Although previous studies show direct impacts of rocky shore disturbance, the impacts can be difficult to predict locally, as the responses of non-target species have spatial and temporal variation (Berthelon *et al.*, 2004). However, the species which are most impacted by physical damage are those which are long lived, sedentary, and slow to reproduce (Berthelon *et al.*, 2004).

Indirect Impacts – Reduced Periwinkle Abundances

Indirect impacts of periwinkle harvesting occur from knock-on impacts of the removal of periwinkles and their activities. Harvesting can alter community interactions, with the impacts dependant on the connection with the non-target species, i.e. the predator, prey, or competitor of *L. littorea* (Quigley, 1999; Berthelon *et al.*, 2004). As a key grazer, as well as prey for birds and crabs, the removal of periwinkles could have large impacts on the whole rocky shore community (Buschbaum, 2000). If periwinkle stocks are impacted by collection, be it reduced abundance or size (Quigley, 1999), the effects of periwinkle grazing and their role as a prey species would be altered. When periwinkles are experimentally excluded from an area of rocky shore, other species have been seen to alter in their abundances (Petraitis, 1989; Cervin and Aberg, 1997; Buschbaum, 2000). Other grazers which compete with *L. littorea,* as well as the algae that periwinkles graze, may benefit from periwinkle overexploitation. Examples of these indirect impacts observed in previous studies can be seen in Table 1:10. Although most of the indirect impacts on non-target species in Table 1:10 appear to be positive if periwinkles were reduced from harvesting, any alterations in a community are not natural when influenced by human activities. Even when impacts are positive for an individual species, the ecosystem as a whole is altered, which goes against the biological conservation ethos of protecting communities from change (Young, 2000).

Impacts on Birds

The effects on shorebirds from intertidal harvesting has been most studied with respect to bait digging. However, many shorebirds utilise the rocky shore as feeding habitats also. The mechanisms of disturbance leading to habitat loss, and reduction in prey species are similar to those described for lugworm collection, only occurring on rocky shores instead. Quigley (1999) observed no effect of periwinkle harvesting

on the three most common bird species on Northumberland rocky shores (Dunlins, Turnstones, and Grey Plover), with higher abundances generally at the collected sites. However, data from the Wetland Bird Survey showed there was lower total abundances of birds at the most visited shore, suggesting that effects are greater on the more ephemeral species (Quigley, 1999).

1.7.3 Summary and Implications for Management

Periwinkles, the rocky shore community as a whole, and birds can be negatively impacted by intertidal harvesting, with impact intensity dependent upon various factors such as: frequency of collection, duration of collection, collection intensity, and nature of the habitat.

Intertidal rocky reefs, where periwinkle collection occurs, are a feature of the BNNC SAC, and as such are protected from degradation (NCAONB, 2009). Activities which negatively impact upon the interest feature need to be managed (MMO, 2014b). This is especially true for birds, being the interest features for the Lindisfarne SPA, and other SPAs falling within the BNNC EMS boundary (many covering rocky shores) (NIFCA, 2013b). Since the overall rocky shore ecosystem has been observed to be impacted by periwinkle collection in previous studies (e.g. Quigley, 1999; Buschbaum, 2000; Berthelon *et al.*, 2004; Crossthwaite, 2012), it is important to research impacts locally within individual protected areas further, and introduce management measures to protect the interest features where appropriate.

The most commonly observed impact of harvesting on periwinkle populations appears to be a reduced average size of individuals (Quigley, 1999; Crossthwaite, 2012). The most appropriate method of management for this impact may be to introduce a minimum harvesting size, which would allow intermediate sizes to thrive (Underwood, 1993), with the aim of maintaining or increasing the average size. Five of the IFCA's currently have a minimum harvesting size regulation in place (Table 1:3), so the potential for NIFCA to do the same within the BNNC EMS is high. However, this method should be tested before further implementation, as it also has the potential to perpetuate the problem with increased targeting of the very largest individuals leading to a smaller average size. Furthermore, in parasitized populations, reduced average size may have a positive effect, as reproductive potential can decrease with increased body size (Robson and Williams, 1970), which should be considered in local management plans.

Overall, periwinkle stocks appear to be relatively resilient to harvesting. The biggest and most worrying potential impacts appear to be those for non-target rocky shore dwelling plants and animals which experience physical disturbance (Berthelon *et al.*, 2004; Crossthwaite, 2012). Since the impacts of boulder turning are more severe when boulders are left upturned (Davenport and Davenport, 2006; AFBI, 2009), management could aim to ensure that collectors return all boulders to their original positions after use, or minimise boulder turning all together. This could be done using education and codes of conduct (Boye *et al.*, 2006). Trampling may be too difficult to manage due to the free access of rocky shores to the public. Many other types of shore users also trample rocky shores, such as rock poolers, anglers, walkers, etc. (JNCC and Natural England, 2011), and so management could not target only intertidal harvesters.

Distinction between commercial and recreational periwinkle collection would be useful for management. Although commercial collection of periwinkles is allowed, being able to manage commercial collection separately could help to reduce the intensity of collection throughout the study area, reducing the severity of impacts. Permitting of commercial collectors could prove a useful management tool (Boye *et al.*, 2006) to limit commercial activity, as could bag or weight limits (Harthill *et al.*, 2005) to control the intensity of commercial harvests.

Overall, there does appear to be potential to reduce the impacts of hand gathering on rocky shores using various management measures. However, more information is needed on the specific impacts within the BNNC EMS to inform managers on the most appropriate methods locally. Data are required on the direct and indirect impacts on both the target species and the associated rocky shore communities within the BNNC EMS.

1.8 Brief Summary and Gaps in Knowledge

The details of intertidal collection activities, such as scale, locale, and intensity, have been little studied to date, the majority of recent studies conducted outside the UK. Within the UK, work has been focussed to the South of England for lugworms (Watson *et al.*, 2015) and Scotland and Ireland for periwinkles (McKay *et al.*, 1997; Cummins *et al.*, 2002). Research in Northern England is lacking. In addition, most studies to date either focus primarily on the ecological impacts, or the social aspect of collection (e.g. how much collection is occurring, when, and where).

Overall, intertidal collection activities are shown within the literature to have negative ecological impacts (e.g., Van den Heiligenberg, 1987; Keough *et al.*, 1993b; Beukema, 1995; Quigley, 1999; Volkenborn and Reise, 2006; Volkenborn *et al.*, 2007a; Volkenborn and Reise, 2007; Hidalgo *et al.*, 2008; Crossthwaite, 2012). This is true for both sandy/muddy and rocky habitat types, and both the target species and associated communities. However the impacts and recovery times depend on many factors, including the intensity of disturbance caused, the length of disturbance, the method of harvesting, and the local habitat and geography (e.g., Blake, 1979a; Cryer *et al.*, 1987; Keough and King, 1991; Keough *et al.*, 1993b; Olive, 1993; Beukema, 1995; Sharpe and Keough, 1998; Berthelon *et al.*, 2004). Inference of impacts of local collection activities based on research conducted in other locations could lead to over or underestimates. Since the details of collection activities vary between geographic locations, the impacts of these activities are also variable. The habitats, conditions, social drivers, and actual harvesting amounts are unique locally, and therefore site specific data is required for effective management. To assess the impacts within the BNNC EMS, local data is required.

There is a shortage of studies which look at both the details of collection (where, when, and how much) and impacts combined, using interdisciplinary methods. An interdisciplinary approach is needed to fully assess intertidal collection activities within the BNNC EMS, gathering data on the activities occurring as well as the impacts they cause.

1.9 Thesis Aims, Objectives, and Structure

This thesis aims to investigate the scale, locale and ecological impacts of the collection of *A. marina, A. defodiens*, and *L. littorea* from shores within the BNNC EMS, using social and ecological research to provide interdisciplinary evidence to inform management. It will achieve this via the following objectives:

1. Quantify the scale and intensity of collection of *A. marina, A. defodiens*, and *L. littorea* within the BNNC EMS
2. Map collection of these species within the BNNC EMS, highlighting hotspots
3. Investigate the current adherence to byelaws/management/rules
4. Investigate the ecological impacts and implications of bait digging for *A. marina* and *A. defodiens* on sandy/muddy shores within the BNNC EMS
5. Investigate the ecological impacts and implications of hand gathering of *L. littorea* from rocky shores within the BNNC EMS
6. Identify areas to prioritise for management and/or monitoring

Table 1:11 and Figure 1:3 show the connections between the objectives and the relevant thesis data chapters, the data used to answer each objective, as well as the order of chapters and the target species each applies to.

Chapter 2 aims to reveal the scale, locale and intensity of both bait digging for lugworms and hand gathering of periwinkles within the BNNC EMS. Annual biomass removal will be estimated, and adherence to byelaws evaluated.

Chapter 3 aims to identify the areas of the BNNC EMS which are most suitable for, sensitive, and vulnerable to lugworm collection activities using spatial modelling techniques.

Chapter 4 aims to investigate the ecological impacts of bait digging for lugworms within the BNNC EMS, by comparing ecological data gathered from shores with a gradient of collection pressures, and by manipulative field experiments.

Chapter 5 aims to investigate the ecological implications of hand gathering of periwinkles within the BNNC EMS, comparing the ecology of shores with differing collection intensities.

Chapter 6 aims to synthesise and discuss the previous chapters' findings in light of management and existing literature.

Table 1:11: Connections between data sources, the objectives they answer, and the chapters they appear in.

Chapter	Objective(s)	Data Sources
2	1, 2, 3, 6	Interviews/questionnaires, shore observations
3	2, 6	Lugworm distribution, literature review, expert opinion
4	4	Sandy shore gradient ecology, simulated digging, lugworm exclusion
5	5	Rocky shore gradient ecology

Lugworm

Periwinkle

Chapter 2: Investigation of the Scale, Locale, and Intensity of Lugworm and Periwinkle Collection Activities within the BNNC EMS

Chapter 3: Modelling the Suitability, Sensitivity, and Vulnerability of the BNNC EMS to Lugworm Collection

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Figure 1:3: Visual summary of data chapters and the target species to which they apply.

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Chapter 2 : Investigation of the Scale, Locale, and Intensity of Lugworm and Periwinkle Collection Activities within the Berwickshire and North Northumberland Coast European Marine Site

2.1 Introduction and Rational

Marine fisheries are still extensively and intensively studied for a wide variety of species and habitats globally (e.g. Couce-Montero *et al.*, 2015; Koslow and Davison, 2016; Mangi *et al.*, 2016; De Wysiecki *et al.*, 2017; Humber *et al.*, 2017; Pauly and Zeller, 2017; Szostek *et al.*, 2017; Thomas *et al.*, 2017; Tiller and Nyman, 2017; Zgliczynski and Sandin, 2017). A significant driver of continued research is the increased recognition of the importance of marine and fisheries conservation in recent years (Soulé *et al.*, 2005; Worm and Branch, 2012). In order to protect marine biodiversity, a global aim (Boonzaier and Pauly, 2016), the details of fishery activities need to be unravelled and understood.

The majority of fisheries stocks exploited globally are classified as data poor (Costello *et al.*, 2012), leading to difficulties in assessing and managing these fisheries (Dowling *et al.*, 2015a). Generally, the statuses of data poor fisheries are thought to be worse than those which are well studied (Worm and Branch, 2012), highlighting the importance of fisheries data for marine conservation. Interest in the development of harvest strategies and fisheries management more generally is increasing, for which fisheries data is essential (Dowling *et al.*, 2015b).

Within the Berwickshire and North Northumberland Coast European Marine Site (BNNC EMS), the need for increased data and understanding of local fishing activities is evident (MMO, 2014b). Intertidal collection from sandflats, mudflats, and rocky shores has been identified as requiring additional information to inform management decisions (MMO, 2014b). This study focusses on two major intertidal fisheries: lugworm and periwinkle.

The understanding of the impacts associated with fisheries are important for management, and as such are well studied (e.g. Auster *et al.*, 1996; Thrush *et al.*, 1998; Turner *et al.*, 1999; Collie *et al.*, 2000; Roy *et al.*, 2003; Masero *et al.*, 2008; Williams *et al.*, 2008; Constantino *et al.*, 2009; Erlandson *et al.*, 2011; Smith *et al.*, 2011; Clarke and Tully, 2014; Hughes *et al.*, 2014; Couce-Montero *et al.*, 2015; Manríquez *et al.*, 2016; Toupoint *et al.*, 2016). However, information on more than impacts alone is needed for successful management and protection, such as knowledge of the harvest effort and estimated catch (Dowling *et al.*, 2015a). Useful empirical indicators for fisheries management include: biomass estimates, catch

rates, and mean size of catch (Dowling *et al.*, 2015a). No such data exists currently for periwinkle or lugworm fisheries within the BNNC EMS.

Over the last 30 years the focus of fisheries management has moved away from top-down, bureaucratic, and science only based approaches, and more onto the importance of involving resource users in the management process (Jentoft *et al.*, 1998). Such newer management approaches include adaptive management (e.g. McLain and Lee, 1996; Berkes *et al.*, 2000; Walters, 2007), ecosystem management (e.g. Link, 2002; Pikitch *et al.*, 2004), and responsible fisheries (e.g. Chakalall and Cochrane, 1996; Cochrane and Chakalall, 2002; Sinclair *et al.*, 2002). Resource users possess knowledge based on experience which can add to fisheries science and improve management (e.g. Jentoft *et al.*, 1998; Berkes *et al.*, 2000; Olsson and Folke, 2001; Wilson *et al.*, 2006; Berkes *et al.*, 2008; Silvano and Valbo-Jørgensen, 2008). Therefore, social methods to gather resource user knowledge and information can be integral in assessing fisheries details.

Spatial information is also important when studying fisheries (Léopold *et al.*, 2014), and spatial management methods are encouraged for marine resources (Hughes *et al.*, 2005; Halpern *et al.*, 2012b). Mapping fisheries can be difficult due to the complex and unstable nature of fisheries over time and space (Stewart *et al.*, 2010), nevertheless distribution data on fishing activity are essential to establish estimates of fishing pressure, understand fishery patterns, and inform management (Stewart *et al.*, 2010; Turner *et al.*, 2015). Despite the importance, quantitative assessment of non-commercial harvests is rare, as the scales of the fisheries do not usually warrant extensive research (Hartill *et al.*, 2005). This often leads to management being based on a scarcity of information, which can lead to inappropriate management measures being implemented (Hartill *et al.*, 2005). Anecdotally, both lugworms and periwinkles are widely collected throughout the BNNC EMS, however, there are no spatial data available on the distribution of collectors.

The majority of existing fisheries studies investigating factors such as fisher distribution, biomass removal, etc. focus on inshore or offshore fisheries, and most often finfish or crustacea (e.g. Gillis *et al.*, 1993; Friedlander *et al.*, 1999; Drinkwater *et al.*, 2006; Stelzenmüller *et al.*, 2008; Bastardie *et al.*, 2010; Bearzi *et al.*, 2010; Cahalan *et al.*, 2015; Natale *et al.*, 2015; Turner *et al.*, 2015). Comparatively few

intertidal fishery studies exist. However, those that do typically use shore observations (both land-based and aerial) and questionnaires or interviews to gather data (e.g. Blake, 1979a; Underwood and Kennelly, 1990; Keough *et al.*, 1993a; McKay *et al.*, 1997; Murray *et al.*, 1999; Cummins *et al.*, 2002; Cunha *et al.*, 2005; Hartill *et al.*, 2005; Carter and Hill, 2007; Sypitkowski *et al.*, 2010; Smallwood *et al.*, 2011; Smallwood and Beckley, 2012; Watson *et al.*, 2015). The often 'black economy' nature of both lugworm and periwinkle collection activities can make gathering fisheries data more difficult, as collectors and wholesalers can be reluctant to communicate with researchers, and there are no official or reliable landings data to refer to (McKay *et al.*, 1997; Cummins *et al.*, 2002). No recent assessments of the details (e.g. scale and intensity) of periwinkle or lugworm harvesting have been undertaken in Northumberland.

Although both lugworm and periwinkle collection is largely unmanaged and unregulated within the BNNC EMS, there are some rules and regulations to consider. Commercial harvesting of lugworms is prohibited (Fowler, 1999), and recreational collection within the BBNC EMS is managed with byelaws at: Boulmer, Newton Haven, and the Lindisfarne National Nature Reserve (NNR) (NCAONB, 2009), as well as a more recent byelaw protecting areas containing seagrass from digging activity (NIFCA, 2013a). Within the Lindisfarne NNR there is a small strip of sandflat either side of the causeway where bait digging is acceptable (UK Marine SACs Project, 2001c). At Boulmer, the northern half of the shore is a no digging zone, whilst no digging at all is allowed at Newton Haven (UK Marine SACs Project, 2001c). Commercial harvesting of periwinkles is allowed (Fowler, 1999), and there are no regulations in place to control the amount harvested (Cummins *et al.*, 2002). Collection can be controlled by fisheries byelaws, such as minimum landing size or closed seasons (Fowler, 1999), however there are currently no byelaws for periwinkle collection within the BNNC EMS.

Commercial fisheries have been repeatedly identified as a major cause of stock declines (e.g. Smith, 2002; Christensen *et al.*, 2003; Pauly *et al.*, 2003), and more recently recreational fisheries have been considered a significant contributor (e.g. Post *et al.*, 2002; Coleman *et al.*, 2004; Cooke and Cowx, 2006). Recreational fisheries are those where fishing is carried out for sport, leisure, or personal consumption (FAO, 1997), whilst commercial fisheries are those where harvests are

for sale (Smith, 2002). Commercial fisheries are often thought to have low effort and high catchability compared to recreational fisheries with high effort and low catchability (Cooke and Cowx, 2006). However, this does not necessarily apply to intertidal fisheries, where the same collection tools and methods are used by both sectors (Fowler, 1999), such as lugworm and periwinkle gathering in Northumberland. Due to high accessibility and limited available space, intertidal commercial and recreational fishers can overlap significantly in both space and time. The direct competition between recreational and commercial fisheries can lead to an increased risk of overharvest (Pereira and Hansen, 2003), and it can be difficult to ascertain the relative contributions of each fishing sector (Griffin, 1988). Therefore, the importance of evaluating commercial versus recreational collection is high if management is to be targeted and successful. Currently, the differentiation between commercial and recreational lugworm and periwinkle collectors is challenging (Watson *et al.*, 2015), which leads to difficulty in assessing and managing the fisheries independently (Watson *et al.*, 2017a).

Compliance of fishery rules is an important issue in marine management, and has been well studied in many fisheries (e.g. Burger *et al.*, 1999; Gezelius, 2002; Crawford *et al.*, 2004; Hatcher and Gordon, 2005; Blank and Gavin, 2009; Bloomfield *et al.*, 2012; Haggarty *et al.*, 2016). Compliance levels can be impacted by various factors, including: the economic gains of non-compliance, deterrence, suitability of the rules, morals of individual fishers, and efficacy of the regulations (Nielsen and Mathiesen, 2003). Bait worm harvesting compliance has been investigated in the Solent, where mixed adherence was observed, varying with location and regulation type (Watson *et al.*, 2015). A lack of enforcement was suggested as the cause of observed non-compliance. The level of compliance or adherence to existing intertidal fisheries rules and regulations within the BNNC EMS is currently unknown, due to the lack of study in this area.

The aim of this chapter is to investigate the scale, locale, and intensity of both commercial and recreational lugworm and periwinkle harvesting within the BNNC EMS. Both spatial and social methods are combined, to explore patterns of collection (including collection method, seasonality, and distribution), and estimate annual biomass removal. Harvesting effort and catch is determined at four rocky (periwinkle) and four sandy (lugworm) shores within the study area over a full annual cycle, and

subsequently extrapolated for the BNNC EMS as a whole. Harvester distribution is mapped across the BNNC EMS, with focus on broad scale positions within the EMS. Additionally, the adherence to current rules and regulations and details of commercial vs recreational collection is explored.

2.2 Methods

Shore observations and questionnaires were used to assess lugworm and periwinkle collection activity within the BNNC EMS, combining two common social methodologies in intertidal fisheries research (e.g. Blake, 1979a; Underwood and Kennelly, 1990; Fairweather, 1991; Kingsford *et al.*, 1991; Keough *et al.*, 1993a; McKay *et al.*, 1997; Murray *et al.*, 1999; Cummins *et al.*, 2002; Cunha *et al.*, 2005; Harthill *et al.*, 2005; Léopold *et al.*, 2014; Watson *et al.*, 2015). Regular observations at selected shores supplied detailed estimates of annual collection activity, whilst single observations from all shores (broad scale observations) were used to extrapolate the estimates over the entire study area. The questionnaires added additional detail to the assessment, and allowed the estimation of annual biomass removal, and differentiation of commercial and recreational collection (Fairweather, 1991).

2.2.1 Regular Shore Observations – Site Selection

Over twenty sediment and rocky shores lie within the BNNC EMS boundaries, of which four were selected for regular observation. These four shores were identified via pilot observations and expert knowledge from collectors and local managers (NIFCA) as sites where either lugworms or periwinkles were collected or were suitable for collection. Both rocky and sandy components were required within a single site so that observations for both target species could be conducted simultaneously. Sites were well spaced throughout the south of the BNNC EMS, driven by suitability and practicality, but allowing for geographical differences between shores. The selected shores were Alnmouth, Boulmer, Newton, and Beadnell (Figure 2:1).

Figure 2:1: Locations of the observation shores in relation to the BNNC EMS as a whole.

Alnmouth beach and estuary (grid reference NU252105) is a large sandy bay joined to the river Aln estuary. Alnmouth rocky shore (grid reference NU258761) is a large outcrop to the north of the beach (Figure 2:2). Collection activity for both target species were unknown for this site.

Boulmer sediment shore (grid reference NU268135) is a medium sized bay composed of muddy sand. Digging is prohibited at the north side of the beach only. To the north lies the substantial Boulmer rocky shore (grid reference NU269101) (Figure 2:3). Both lugworm and periwinkle collection were known to occur here anecdotally.

The sediment shore at Newton (grid reference NU241245) is a large sandy, sheltered bay, were digging is prohibited. The rocky shore component (grid

reference NU244312) was small outcrops split either side of the sandy bay (Figure 2:4). Lugworm collection was known to occur here anecdotally, whilst periwinkle activity was unknown.

Beadnell sediment shore (grid reference NU233283) is very large and sheltered. The rocky shore (grid reference NU233432) to the north is equally large, stretching along a significant length of the coast (Figure 2:5). Periwinkle collection was known to occur here anecdotally, whilst lugworm activity was unknown.

Figure 2:2: Alnmouth A) Aerial image showing both rocky and sandy shore elements (Map data @2018 Google). B) Photograph of rocky shore. C) Photograph of sandy shore.

Figure 2:3: Boulmer A) Aerial image showing both rocky and sandy shore elements (Map data @2018 Google). B) Photograph of rocky shore. C) Photograph of sandy shore.

Figure 2:4: Newton A) Aerial image showing both rocky and sandy shore elements (Map data @2018 Google). B) Photograph of rocky shore. C) Photograph of sandy shore.

Figure 2:5: Beadnell A) Aerial image showing both rocky and sandy shore elements (Map data @2018 Google). B) Photograph of rocky shore. C) Photograph of sandy shore.

2.2.2 Regular Shore Observations

Observation methods were piloted in October and November 2014. Shores were visited for four hours (two before low water, two after low water) on a mixture of spring and neap tides, day and night hours, and weekdays and weekends. The pilot visits revealed that night time observations were necessary for lugworm collection,

but only on spring tides, when the lugworm beds were exposed enough to warrant more difficult night time collection (no night time collection observed on neap tides). Additionally, it was concluded that a shorter observation time close to low water was sufficient to record the number of collectors, due to all collectors observed being present at low water despite varying collecting times and patterns.

Regular shore observations began in December 2014, and ran for twelve months, to capture seasonal differences (Fowler, 1999). Each of the four sites (eight shores – one rocky and one sandy at each site) were observed at low water six times each month, totalling 288 observations per habitat type and target species. Each of the six monthly observations were categorised as: Spring Day Weekday, Spring Day Weekend, Spring Night Weekday, Spring Night Weekend, Neap Day Weekday, and Neap Day Weekend. These categories were designed to capture variation, standardise observations between shores, and allow subsequent extrapolation of results over unobserved days (Cunha *et al.*, 2005). The differentiation between spring and neap tides was required due to the belief that more collection would occur on spring tides when more of the target species are exposed (Fowler, 1999). Weekends and weekdays were separated to account for working patterns of non-commercial collectors, and day and night tides split on the recommendation of Underwood and Kennelly (1990).

At each observation visit the number of collectors present on each shore was recorded at the time of low water. Binoculars were used to observe from a distance, with the purpose of recording natural behaviour. Day time observations also recorded the method of collection for lugworm harvesting (fork or bait pump (Fowler, 1999)), the adherence to byelaws, and for September, October, and November 2015 the positions of collectors within the sediment shores (Kingsford *et al.*, 1991). Night observations recorded the number of head torch lights visible on the shore.

2.2.3 Broad Scale Observations

To relate the detailed observations of four shores described above to the BNNC EMS as a whole, one-off broad scale observations were conducted for each of the target species. On selected days multiple volunteers aimed to observe as many shores as possible at low water. This allowed comparisons between the four targeted shores and others across the BNNC EMS.

Observations were conducted on the 24th January 2015 for lugworms, and 2nd August 2015 for periwinkles. Dates were selected to maximise the number of collectors observed. The lugworm date was at a weekend in peak winter bait digging season (Fowler, 1999), one day before a major local fishing competition (Amble open), with a spring low tide falling in late morning. The periwinkle date was at a weekend in summer (when more collection was observed), with a spring low tide late morning. Fifteen sediment shores were observed at the same time (low water) on each date. The observation sites were spread along the entire BNNC EMS coastline, with sites at both the northern and southern boundaries, and an even spread throughout. The observed shores are shown in Figure 2:6.
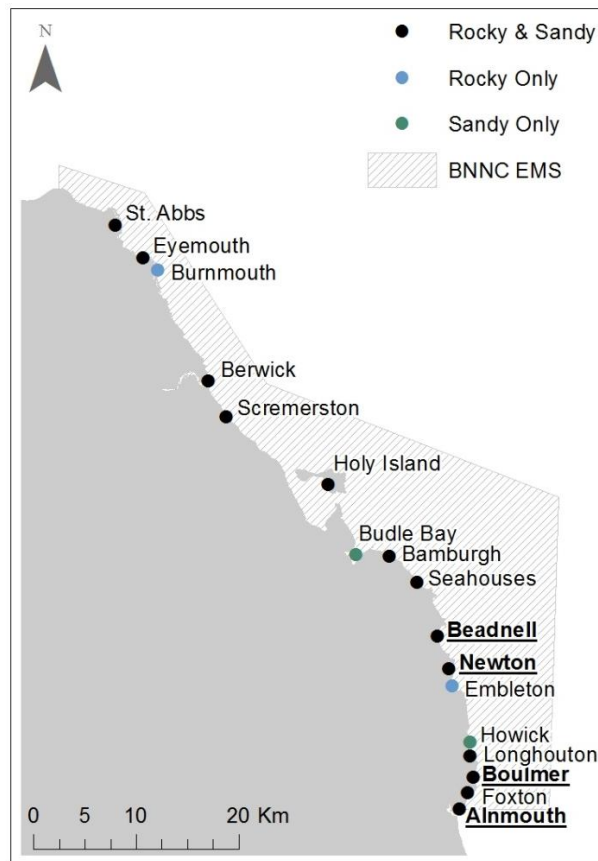
Figure 2:6: Locations of the broad scale observation shores within the BNNC EMS, with the regular observation shores bold-underlined. Sediment shores (black and green circles) were observed for lugworm collection on 24th January 2015. Rocky shores (black and blue circles) were observed for periwinkle collection on 2nd August 2015.

Volunteer pairs made the observations so that low water could be observed accurately at each location (requiring a lot of people spaced along the coast). For stretches of shores (for example Longhoughton and Howick) where sediment or rocky shores continued intermittently over long distances, volunteers cycled the length of the shore, observing from coastal paths. Methods were the same as regular shore observations, so results are comparable. Volunteers were trained for consistency prior to observations. This included identifying collectors, and recognising the collection methods for lugworms. The observation details for both days can be seen in Table 2:1.

Table 2:1: Details of the broad scale observations days.

Target Species and Date	Shore	Observer Pair ID	Transport Method Between Sites	Time of Low Water	Approx. Observation Start Time
Lugworm - 24th January 2015	Alnmouth	A	Walking	11:50am	11:45am
	Foxton	A	Walking	11:50am	12:00pm
	Boulmer	B	Cycling	11:50am	11:40am
	Longhouton	B	Cycling	11:45am	11:50am
	Howick	B	Cycling	11:45am	12:05pm
	Newton	C	Driving	11:25am	11:40am
	Beadnell	D	Driving	11:10am	11:00am
	Seahouses	D	Driving	11:10am	11:20am
	Bamburgh	E	Driving	11:05am	11:15am
	Budle Bay	E	Driving	11:05am	11:00am
	Holy Island	F	N/A	11:00am	10:50am
	Scremerston	C	Driving	11:00am	11:00am
	Berwick	C	Driving	11:00am	10:45am
	Eyemouth	H	Driving	11:00am	10:55am
	St Abbs	H	Driving	11:00am	11:10am
Periwinkle – 2nd August 2015	Alnmouth	I	Walking	11:30am	11:20am
	Foxton	I	Walking	11:30am	11:35am
	Boulmer	J	Cycling	11:30am	11:20am
	Longhouton	J	Cycling	11:25am	11:35am
	Embleton	K	Walking	11:10am	11:15am
	Newton	K	Walking	11:10am	11:00am
	Beadnell	L	N/A	10:50am	10:50am
	Seahouses	M	Driving	10:50am	11:00am
	Bamburgh	M	Driving	10:50am	10:40am
	Holy Island	N	N/A	10:45am	10:45am
	Scremerston	O	Driving	10:45am	10:55am
	Berwick	O	Driving	10:45am	10:40am
	Burnmouth	P	Driving	10:45am	10:35am
	Eyemouth	P	Driving	10:45am	10:45am
	St Abbs	P	Driving	10:45am	11:00am

2.2.4 Questionnaires

A social survey was designed to gather more detailed information on the intensity and nature of collection activities, including exploring differences in commercial and recreational activities. Questions were intended to determine factors such as collection hotspots, frequency and duration of collecting trips, seasonal collection patterns, and the number or mass of the target species removed per trip.

Surveys were administered face to face, during shore observation trips. This method involves synchronous communication, allowing social cues to be recognised, and resulting in spontaneous and non-reflective responses (Opdenakker, 2006). To increase responses, the lugworm collection questionnaire was also available online using 'smartsurvey.co.uk', and distributed via a link which was shared on relevant social media pages, and a well-known angling forum (NESA). The periwinkle questionnaire was not promoted online, due to the lack of a central base to make contact. Mixed results were obtained online, where negative responses from anglers and bait diggers became evident. The response from face to face surveys was more positive, with the majority of lugworm collectors approached willing to participate. Periwinkle collectors were more reluctant to talk, which together with fewer encounters on shore observations, resulted in significantly less responses. Negative responses, and unwillingness to participate were to be expected due to the black market nature of intertidal collection, as has been reported in previous studies (e.g. Cummins *et al.*, 2002).

Issues of questionnaire methods are well recognised, especially in relation to the fidelity of answers for sensitive or threatening topics and questions (Bradburn *et al.*, 1979; Rasinski *et al.*, 1999; Tourangeau and Yan, 2007). Despite the secrecy often involved in intertidal collection activities, this studies questionnaire does not ask any extremely sensitive questions, and respects the respondents privacy by not asking for personal details (e.g. name, age, gender, home town, etc.), an important consideration of survey design (Rasinski *et al.*, 1999). Additionally, respondents were not asked directly if they were commercial collectors, with the topic only discussed if voluntarily brought up, as this is a sensitive question for many collectors. It is reported that in-depth interviews generally have more honest answers given than self-completion questionnaires and face to face questionnaires in social studies

with sensitive topics (Plummer *et al.*, 2004). However, this in-depth method was not deemed necessary for use within this study, and resources were better suited to gathering a higher number of shorter and focused questionnaire responses.

For further analysis of commercial vs recreational collection details, the responses were separated into expected commercial and expected recreational categories based on similarities or differences with several self-confessed commercial collectors.

2.2.5 Estimating Biomass Removal

Total annual harvests of lugworms and periwinkles (kg) were calculated using separate estimates of harvesting effort and harvest rate (Cunha *et al.*, 2005). Harvesting effort was ascertained via shore observations, and harvesting rate via social survey (both described above). The regular shore observations provided the number of collectors for each designed category (month, neap/spring, week/weekend, and day/night – e.g. January Neap Day Weekend). Categories were based on knowledge that seasonality, tidal state, day of the week, and time of day/night could all influence harvesting effort (Cunha *et al.*, 2005). The questionnaires provided the mean number of lugworms and the mean mass of periwinkles removed per collector per trip. For lugworms, the mean mass harvested per collector per trip was subsequently calculated using the mean mass of 50 lugworms collected from Boulmer (6g).

Once mean mass of lugworms and periwinkles removed per observation was calculated (product of mean mass harvested per collector per trip and number of collectors observed), totals for each category as a whole could be estimated. For this, the number of low tides falling within each category needed to be known (Table 2:2), so that the mean mass for a single observation could be multiplied by the number of similar low tides, resulting in the extrapolation of data over all non-observed low tides, giving the mean mass removed per observation category. This was calculated for all 72 categories (6 per month).

Table 2:2: The number of low tides in each category per month. Neap ≥ 1.0m low tide (or ≥ 1.15m in Nov & Dec). Night = hours of darkness.

Number of low tides in each observation grouping

	J	F	M	A	M	J	J	A	S	O	N	D
Neap Day Weekend	7	6	5	5	8	5	4	4	4	7	6	6
Neap Day Week	18	16	17	15	11	13	14	12	13	11	14	17
Spring Day Weekend	2	2	4	3	2	3	4	6	4	2	3	2
Spring Day Week	4	4	5	7	10	9	10	9	9	11	7	5
Spring Night Weekend	2	3	4	4	3	3	3	2	1	3	4	4
Spring Night Week	6	10	10	11	9	7	4	7	6	5	7	9

The estimates of mean mass removed per category were summed to give the total annual removal estimate from the observed shores. Ratios gained from the broad scale shore observations (described above) were used to extrapolate the data further to include harvesting from the unobserved shores within the BNNC EMS. Resulting in an estimate of annual biomass removal for the BNNC EMS as a whole. Visualisation of the order of calculations and the data required for each step of the biomass estimate can be seen in Figure 2:7. Biomass estimates were subsequently converted to economic value using average retail prices from the literature and shellfish wholesalers (The Fish Society, 2017; Watson *et al.*, 2017a).
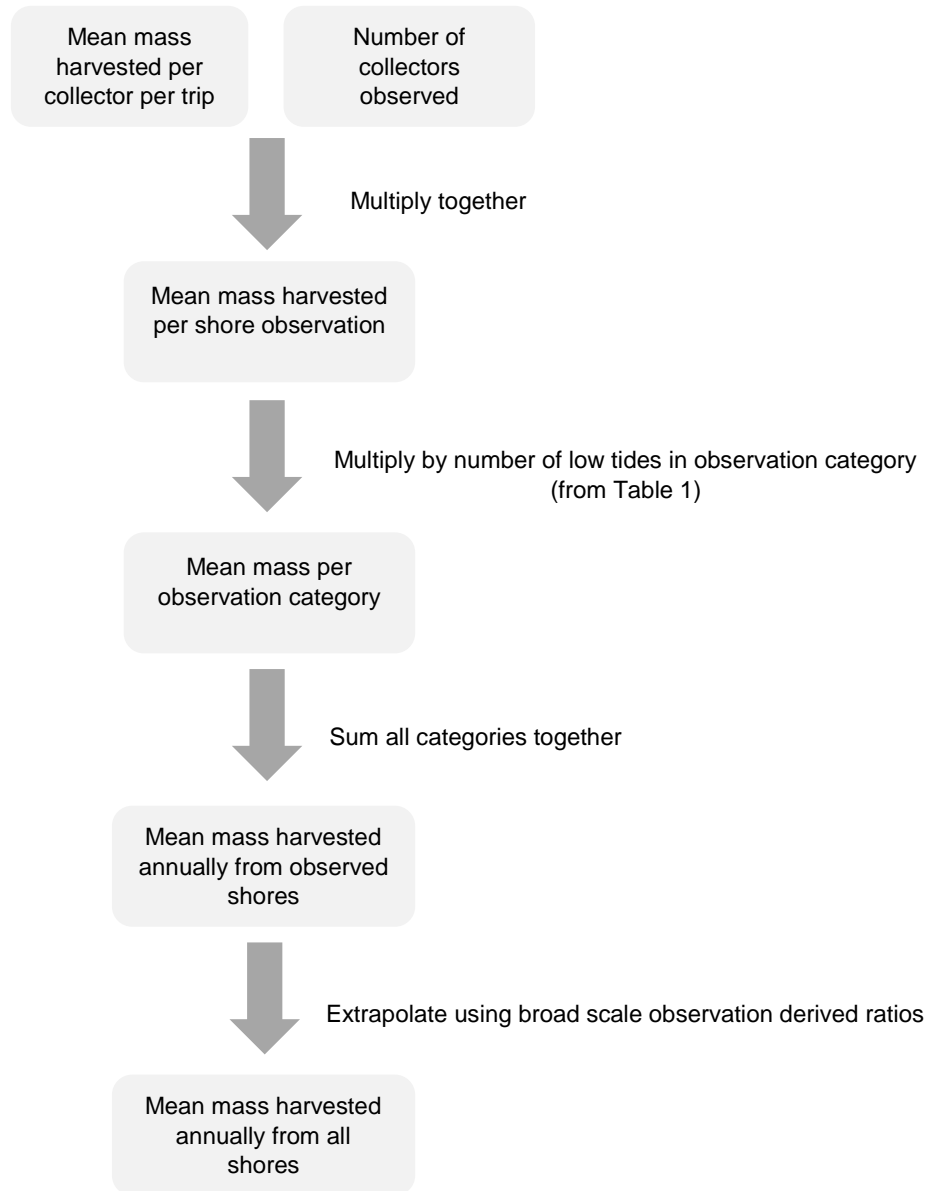
Figure 2:7: Flow chart showing the order of calculations used to estimate the annual biomass removal from the BNNC EMS for both periwinkle and lugworm harvesting, and the data required for each stage.

2.2.6 Data Analysis

Minitab version 17 was used to analyse differences in collection observed between shores and observation categories. Data were zero inflated and did not conform to normal distribution (Kolmogorov Smirnov, $P < 0.05$), so non-parametric analyses were used – Kruskal-Wallis and Mann-Whitney (Underwood, 1997; Dytham, 2011). Questionnaire data were also analysed using non-parametric tests due to non-

normal distribution and unequal sample sizes between groupings. ArcMap GIS software was used for mapping collector distributions and densities.

2.3 Results

2.3.1 Regular Shore Observations

Over 12 months of regular shore observations, a total of 241 lugworm and 62 periwinkle collectors were witnessed on the shores. The majority of both lugworm and periwinkle collectors were observed at Boulmer. A high number of lugworm collectors were also observed at Newton, with a single sighting at Alnmouth, and none at Beadnell. For periwinkles, Beadnell was a popular collection shore, while Newton and Alnmouth had a lower level of collection. The total number of collectors observed at each shore can be seen in Figure 2:8. The average number of collectors recorded per observation were statistically different between shores for both lugworm and periwinkle collectors (Kruskal-Wallis, H = 97.91, 13.35, df = 3, 3, *P* < 0.001, 0.01), demonstrating clear shore preferences for both target species.
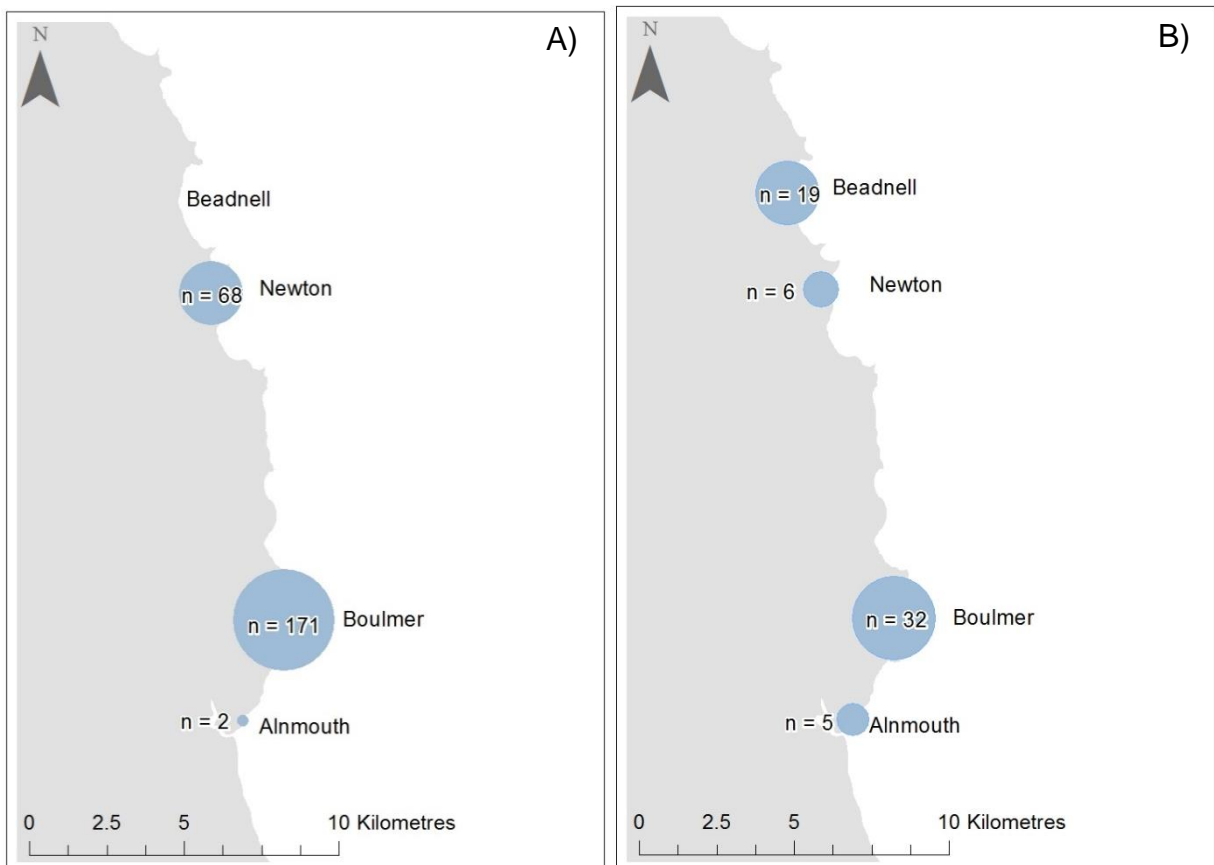
Figure 2:8: Total number of collectors observed per shore during 12 months of regular observations. A) Lugworm collectors on sediment shores. B) Periwinkle collectors on rocky shores. n = total number of collectors recorded.

Seasonal effects are strong for both fisheries, but patterns are opposite. Lugworm collection occurred mainly in winter, with peak numbers observed in January and February (Figure 2:9 A). Periwinkle collectors were most active in summer, with August being the most collected month (Figure 2:9 B). These seasonal patterns were consistent over all observed shores.
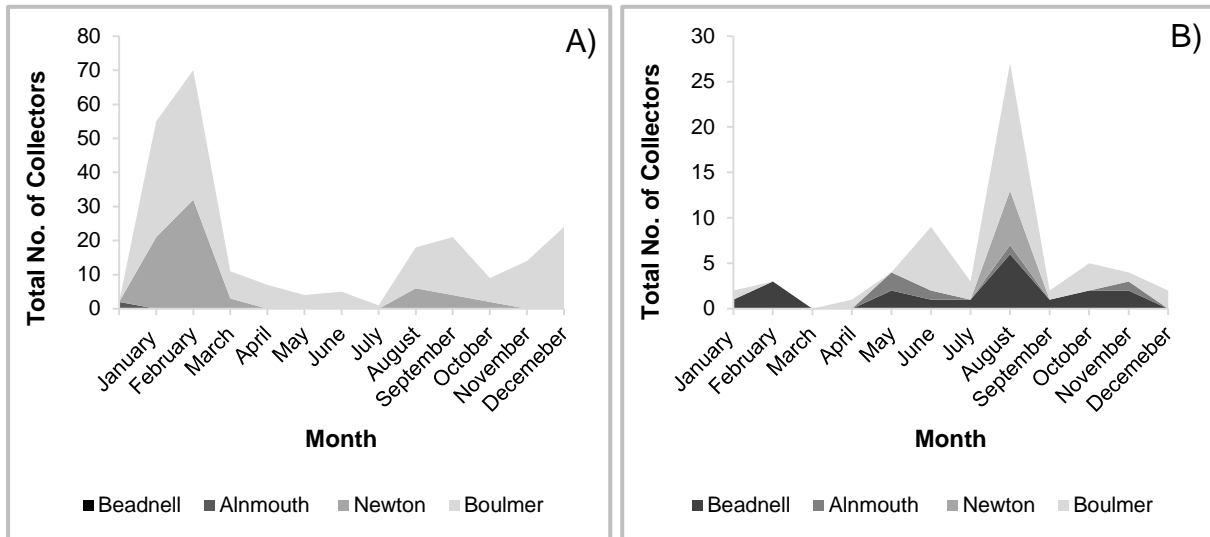
Figure 2:9: Total number of collectors observed each month at each regularly observed shore. A) Lugworm collectors on sediment shores. B) Periwinkle collectors on rocky shores.

Tidal state also affected the number of collectors observed. Spring tides attracted more lugworm and periwinkle collectors than neap tides, with 92% of daytime lugworm collectors observed on spring tides, and 69% of periwinkle collectors. The average number of collectors recorded per observation was statistically different between spring and neap tides for both lugworm and periwinkle fisheries (Mann-Whitney U-test, $U = 10317.5, 9836$, $n_{spring, neap} = 96$, $p < 0.0001, 0.02$), demonstrating a preference for spring tide collection.

No periwinkle collection was recorded during night observations. Lugworm collection was considerable during night observations at both Boulmer and Newton. Overall, the number of collectors observed at day and night observations were similar. However, the prevalence of night collection varied between shores. Within spring tide observations, 36% of Boulmer and 81% of Newton collector recordings fell within night observations. Collectors at Newton clearly have a stronger preference for night tides (Mann-Whitney U-test, $U = 1662.5$, $n_{day, night} = 48, 24$, $p < 0.052$). Both digging

fork and bait pump collection methods were used by lugworm collectors during observations. The fork method proved to be significantly more popular (Mann-Whitney U-test, U = 39363.5, $n_{fork, pump}$ = 192, p < 0.0005), with 85% of day time collectors using this method.

Byelaw adherence varied between locations. All lugworm collectors at Newton were in breach of the 'no digging' byelaw which covers the entire lower shore. Boulmer byelaw was in contrast relatively well adhered to. Collectors were regularly close to the boundary of the 'no digging zone', but overall only 9 were observed fully inside the prohibited area during daytime observations (5% of collectors). Of this, only 2% of total collectors were actually digging in this zone, with the others using bait pumps (which can be argued are not covered by the legislation).

Within shore lugworm collector distribution was recorded for 3 months (September, October, and November 2015) at Boulmer to identify small scale hotspots and further examine byelaw adherence. Figure 2:10 displays the locations of each collector observed during daytime observations during this period. The majority of collectors were situated within the southern half of the shore and close to low water, whilst only four collectors (all using bait pumps) were recorded within the 'no digging zone'.
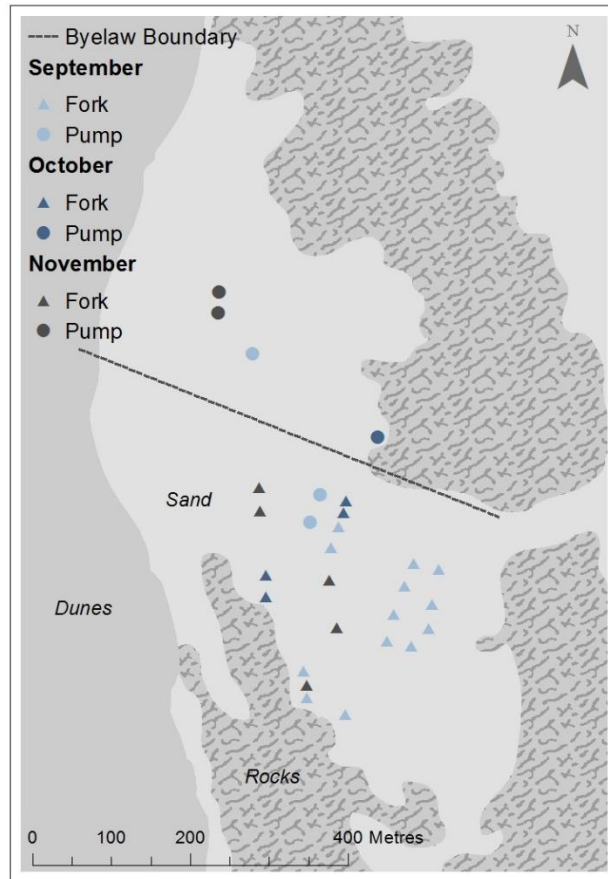
Figure 2:10: Within shore lugworm collector distribution at Boulmer. Different colours represent the three observation months – September, October, and November 2015.

2.3.2 Broad Scale Observations

The regularly observed shores contained a significant amount of the collection activity recorded during the broad scale observations, suggesting they were appropriate choices for the regular shore observations. The presence of lugworm collectors at Alnmouth and periwinkle collectors at Newton on the broad scale observation days (despite no recordings on other regular observations) confirmed that the chosen observation dates maximised the sightings as planned.

New collection hotspots identified include Berwick for lugworm collection, and Berwick, Bamburgh, and Seahouses for periwinkle collection (Figure 2:11 B). The most popular collection shore was Boulmer for lugworms, and Seahouses for periwinkles (Figure 2:11 A). Periwinkle collection was distributed more widely over the study area than lugworm collection. Only five of the fifteen observed shores contained lugworm collectors, compared to nine for periwinkle collectors, suggesting

that shore selection or suitability is more important when harvesting lugworms. In both cases, collector distribution was skewed to the south of the BNNC EMS.
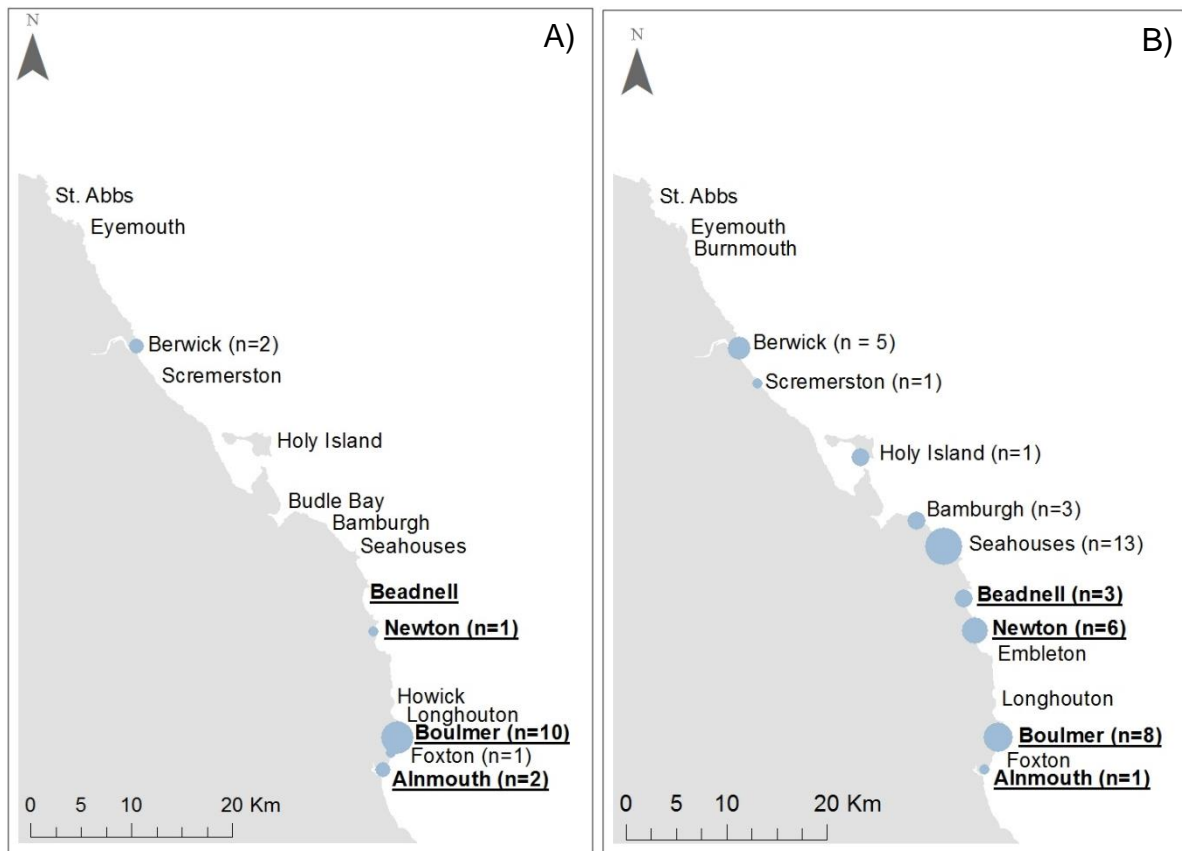
Figure 2:11: Number of collectors observed per shore during the broad scale observation days. A) Lugworm collectors on sediment shores (24th January 2015). B) Periwinkle collectors on rocky shores (2nd August 2015). Regularly observed shores in bold underlined. n = total number of collectors.

Ratios were developed for newly identified collection sites compared to regularly observed shores, for subsequent use in BNNC EMS wide biomass estimates. Boulmer was chosen as the standard for regularly observed shores (ratio of one), due to the popularity for both target species collection. All regularly observed shores were not given a ratio due to more detailed and accurate data being available. Shores with no collectors recorded on the broad scale observation days were assumed to generally have no collection, and are given a ratio of zero. The resulting ratios for all shores can be seen in Table 2:3.

Table 2:3: Ratios (1 d.p) of collector numbers for the broad scale collection day shores when compared to Boulmer collection levels.

	Alnmouth	Foxton	Boulmer	Longhought	Howick	Embleton	Newton	Beadnell	Seahouses	Bamburgh	Budle Bay	Holy Island	Scremerston	Berwick	Burnmouth	Eyemouth	St Abbs
Lugworm	-	0.1	1	0	0	0	-	-	0	0	0	0	0	0.2	0	0	0
Periwinkle	-	0	1	0	0	0	-	-	1.6	0.4	0	0.1	0.1	0.6	0	0	0

2.3.3 Questionnaires – Collection Details

A total of 66 questionnaire responses were received from lugworm collectors, of which 27 were online, and 39 face to face. Periwinkle collection responses totalled 11, all face to face.

Lugworm Collection questionnaire responses supported the patterns observed during regular and broad scale shore observations. The most popular collection shores were Boulmer, Berwick, and Newton. All other shores included in the questionnaire had at least one respondent declaring that collection occurs there, with one stating that "*collection occurs everywhere there are lugworms*". Collection is likely to occur on most shores within the BNNC EMS where lugworm density is high enough for effective collection, whilst higher collection intensity appears to occur at a few main shores. Seasonality of collection supported that recorded from shore observations. The majority of collectors (78%) only harvest lugworms in winter, namely September through February. However, several collectors harvest year round, presumably related to commercial collectors maintaining a regular income.

Weekends were the preferred collection day, with 88% of respondents collecting on Saturdays and Sundays compared to 56% on weekdays. The low tide height (i.e. spring vs neap tides) was a consideration for 83% of respondents when deciding when to harvest lugworms, with spring tides being the preferred condition. Around half the respondents collect lugworms in hours of darkness, be that early mornings, late evenings, or middle of the night. Digging with a fork was the most popular harvesting method, with 70% of people using this method either alone or combined with a bait pump. The details of collection trips varied substantially between respondents. Harvesting frequency ranged from every other day to every few

months. Respondents spent between less than an hour and four hours collecting per tide, and harvested between less than 50 and more than 700 worms each time. The majority of collectors harvested less than 200 worms per trip (82%), with a mean of 135.22 (±143.78 SD) worms.

Choosing a harvesting location was mainly based on lugworm size and density for the majority of collectors. 93% selected lugworm density as an important factor, and 71% consider lugworm size important. Other popular consideration factors included travel distance from home (37%), sediment type (27%), and parking availability (15%).

Several rocky shores were identified as harvesting locations for periwinkles by the questionnaire respondents: Boulmer, Alnmouth, Newton, Howick, Beadnell, Berwick, and Eyemouth. Summer months were the preferred collection period (May, June, July, and August), and none of the respondents collect periwinkles in hours of darkness. Collection details varied greatly between respondents. Frequency of collection ranged from daily to every few months. Collection periods last between 1 and 5 hours, and respondents harvest between a few pounds and 7 stone per trip. Most collectors harvested less than 20 pounds per trip (55%), with a mean value of 30.59 (±25.24) pounds.

2.3.4 Questionnaires - Commercial vs Recreational

There were clear differences in the questionnaire responses between expected commercial collectors and recreational collectors. Commercial collectors for both lugworms and periwinkles generally harvested larger amounts per trip, spent longer collecting per trip, and collected more often (Table 2:4 and Table 2:5). The differences between commercial and recreational lugworm harvester responses were statistically significant for the number of trips per month (Mann-Whitney U-test, $U = 472.5$, n commercial, recreational $= 9, 57$, $P < 0.001$), hours spent collecting per trip (Mann-Whitney U-test, $U = 501.5$, n commercial, recreational $= 9, 57$, $P < 0.001$), and the number of worms harvested per trip (Mann-Whitney U-test, $U = 538.0$, n commercial, recreational $= 9, 57$, $P < 0.001$). The differences between commercial and recreational periwinkle harvester responses were also statistically significant for the number of trips per month (Mann-Whitney U-test, $U = 41.0$, n commercial, recreational $= 5, 6$, $P < 0.05$), hours spent collecting per trip (Mann-Whitney U-test, $U = 41.5$, n commercial, recreational $=$

5, 6, P < 0.05), and the mass of periwinkles harvested per trip (Mann-Whitney U-test, U = 45.0, n commercial, recreational = 5, 6, P < 0.05).

Table 2:4: Lugworm collection details (means ± SD) for all collectors combined (n = 66), commercial only (n = 9), and recreational only (n = 57).

	No. of Trips per Month	Hours Spent per Trip	No. of Worms per Trip
Combined	3.15 (±3.61)	2.17 (±0.85)	135.22 (±143.78)
Commercial	8.67 (±6.28)	3.28 (±0.67)	400.00 (±196.85)
Recreational	2.28 (±1.95)	1.99 (±0.73)	94.30 (±71.81)

Table 2:5: Periwinkle collection details (means ± SD) for all collectors combined (n = 11), commercial only (n = 5), and recreational only (n = 6).

	No. of Trips per Month	Hours Spent per Trip	Mass Collected per Trip (lbs)
Combined	4.78 (±8.68)	2.86 (±1.12)	30.59 (±25.24)
Commercial	9.00 (±12.04)	3.50 (±0.71)	51.80 (±23.00)
Recreational	1.27 (±1.38)	2.17 (±0.82)	12.92 (±5.10)

Using the means from Table 2:4 and Table 2:5 to calculate the average amount collected per person per month results in 215 worms per recreational lugworm collector, 3,468 worms per commercial lugworm collector, 16.41 lbs per recreational periwinkle collector, and 466.20 lbs per commercial periwinkle collector. The average amount collected monthly by individual commercial collectors for both lugworm and periwinkle collection is far higher than that of recreational collectors, as much as 16 and 28 times higher respectively. If ratios of commercial to recreational collectors from the questionnaire respondents are assumed to be representative of the industries within the BNNC EMS as a whole, then commercial lugworm collectors are estimated to take 72.8% of the harvested worms, and commercial periwinkle collectors 95.9% of the harvested periwinkles.

2.3.5 Biomass Removal Estimates

Annual biomass estimates were calculated using the number of collectors per shore from regular shore observations and the mean mass harvested per collector per trip from questionnaires. The mean number of worms collected per trip was 135.22, with an average worm mass 0f 6.0g, resulting in an estimated average biomass of 0.81 kg of lugworms harvested per collector per trip. High and low scenarios were also considered using 95% CIs of the number of worms collected, resulting in an estimated 0.60 – 1.02 kg. The average mass of periwinkles collected per person per

trip was 13.87 kg. High and low scenarios for periwinkle harvesting mass are 7.11 – 20.64 kg.

Data were extrapolated over all unobserved days using the number of days in each observation category (see 2.2.3 Estimating Biomass Removal – Table 2:2) and further extrapolated onto all unobserved shores using broad scale observation ratios (see 2.3.2 Broad Scale Observations - Table 2:3). The annual biomass removal estimates for each identified collection site and the BNNC EMS as a whole can be seen in Table 2:6 for lugworms and Table 2:7 for periwinkles, with average, high, and low scenarios for each. The average estimates are 1.24 tonnes of lugworms and 13.40 tonnes periwinkles removed annually from shores lying within the BNNC EMS boundaries. However, low and high scenarios (95% CIs) suggest values could lie between 0.92 and 1.56 tonnes for lugworm, and between 6.86 and 19.93 tonnes for periwinkle harvests.

Based on the average bait worm UK retail value of £42 per Kg (Watson *et al.*, 2017a), the lugworm fishery in the BNNC EMS is estimated to be worth £52,128, with low and high scenarios of £38,747 and £65,509. The periwinkle fishery is estimated at £133,982, based on an average retail value of £10 per Kg (Berwick Shellfish Company, 2017; The Fish Society, 2017), with low and high scenarios of £68,633 and   £199,330.

Table 2:6: Total annual number of lugworm collectors (rounded to whole numbers when using ratios) visiting each sediment shore, and corresponding biomass removal estimate (kg, 2 d.p) for each collected shore within the BNNC EMS, and the area as a whole.

Shore	No. of Collectors	Average Biomass Removed (kg)	Low Scenario Biomass Removed (kg)	High Scenario Biomass Removed (kg)
Alnmouth	4	3.25	2.41	4.08
Boulmer	876	710.72	528.28	893.15
Foxton	88	71.07	52.83	89.32
Newton	387	313.98	233.38	394.58
Berwick	175	142.14	105.66	178.63
BNNC EMS	**1,530**	**1,241.16**	**922.56**	**1,559.75**

Table 2:7: Total annual number of periwinkle collectors (rounded to whole numbers when using ratios) visiting each rocky shore, and corresponding biomass removal estimate (kg, 2 d.p) for each collected shore within the BNNC EMS, and the area as a whole.

Shore	No. of Collectors	Average Biomass Removed (kg)	Low Scenario Biomass Removed (kg)	High Scenario Biomass Removed (kg)
Alnmouth	24	333.01	170.59	495.43
Boulmer	191	2,650.20	1,357.59	3,942.81
Newton	36	499.51	255.88	743.15
Beadnell	139	1,928.68	987.98	2,869.37
Seahouses	310	4,306.57	2,206.08	6,407.06
Bamburgh	116	1,614.96	827.28	2,402.65
Holy Island	15	207.05	106.06	308.03
Scremerston	15	201.87	103.41	300.33
Berwick	119	1,656.37	848.49	2,464.25
BNNC EMS	**1,530**	**13,398.22**	**6,863.36**	**19,933.09**

2.4 Discussion

2.4.1 Shore Observations

Lugworm collectors were observed more frequently than periwinkle collectors during regular shore observations, likely due to the large, sustained, angling community which both drives and carries out lugworm collection (Angling Trust, 2013), and the decreasing, aging, population of periwinkle collectors observed in other locations (Cummins *et al.*, 2002).

There were clear site preferences for both lugworm and periwinkle collection activities during regular shore observations. Broadscale observations revealed that site selectivity was greater for lugworm than periwinkle collection, with fewer collection sites identified. Many factors play a role in deciding how suitable and therefore popular a site is for certain activities (Phillips and House, 2009; Paudel *et al.*, 2011; MMO, 2012). Key considerations include: ease of access, travel distance, safety, target species abundance, etc. (e.g. Phillips and House, 2009; Paudel *et al.*, 2011; Villamagna *et al.*, 2014). It is thought that personal taste, as well as site characteristics can play a part (Paudel et al., 2011).

Boulmer and Newton reamained popular lugworm collection shores despite legislative restrictions (NCAONB, 2009), possibly because they cointain both *Arenicola marina* and the scarcer *Arenicola defodiens* (personal observation) which is often favoured by anglers (Fowler, 1999). Broad scale observations identified Seahouses as a very popular periwinkle collection site, which is a recognised commercial location (Northumberland County Northumberland County Council, 2014a). Berwick was also popular, which was expected due to the close proximity of a major shellfish wholesaler (Berwick Shellfish Company) for fast and convenient sales.

There was a southerly skew to the collection sites identified during broad scale observations for both collection activities. This skew can be explained by human population distribution within the study area. North Northumberland has a population density of 26.3 people per km^2, compared to South East Northumberland with 737 people per km^2 (Northumberland County Northumberland County Council, 2014b). South East Northumberland's northern border lies just below the BNNC EMS. It is possible that many of the collectors in the south of the EMS travel from the densely

populated South East Northumberland. Informal interviews around questionnaires confirmed this, with many collectors regularly travelling from as far south as Sunderland to collect lugworms from Boulmer.

Seasonality was strong for both collection activities. The majority of lugworm harvesting occurred in winter, which was expected due to higher bait demand from the winter fishing season - specifically Cod fishing (Townshend and O'Connor, 1993). In contrast, summer was the peak periwinkle collection season. This was once the low season for periwinkle sales, however the introduction of exporting to Europe has since increased the summer demands (Cummins *et al.*, 2002), allowing summer commercial collection to thrive.

Spring tides were favoured by both collection activities, which is in line with previous studies (Cummins *et al.*, 2002). During spring tides a larger area of shore is exposed and available for harvesting. In addition to more stock available, the body size of both target species are generally thought to increase lower down the shore (Chapman and Newell, 1949; Bruce *et al.*, 1963; Perez *et al.*, 2009), allowing for better quality and higher value harvests (Cummins *et al.*, 2002). Additionally, *A. defodiens* is only exposed by the lowest tides (Fowler, 1999), so bait pumping can only occur at these times. Neap tide collection did occur, but with a much lower frequency and intensity. Some of these lugworm collectors were known commercial operators whom collected at all tidal states despite conditions not being ideal.

Digging was likely more popular than pumping because it is more flexible - capable of harvesting both *A. marina* and *A. defodiens,* at a variety of tidal states. Conversations with collectors revealed that pumps can be difficult to use and require a specific technique, which many stated they gave up on after unsuccessful collection attempts: "bait pumps can save your back, but they are faffy and I can't get the knack, so I don't use mine anymore" (questionnaire respondent, personal communication, 2015). The flexibility as well as ease of use may explain the popularity of the traditional digging fork. Bait pumps create far less sediment disturbance during collection, and are thought to have a lower impact on other infaunal invertebrates (Fowler, 1999). If bait pump proportional use was to increase in the future, some of the negative impacts of lugworm collection (e.g. reduction in

infaunal invertebrate abundance and species richness (e.g. Van den Heiligenberg, 1987; Beukema, 1995; Brown and Wilson, 1997)) may be reduced.

Night collection was only recorded for lugworms, with substantial collection occurring at both Boulmer and Newton during hours of darkness. This is in contrast to observations of bait digging activity in the Solent, where night collections were less common (Watson *et al.*, 2015). Many of the lowest spring tides occurred at night during the observation period, which explains the willingness of many collectors to harvest in these conditions. The vast majority of Newton collection occurred in hours of darkness, with a lower but still significant proportion at Boulmer. The higher night collection at Newton is likely due to collectors avoiding enforcement of the bait digging legislation in place there. National Trust rangers or wardens ask bait diggers to leave the shore when observed (personal communication). No patrols occur at night, leaving the shore open for collection without enforcement. When detection probability is low, illegal fishing is more likely to occur (Nielsen and Mathiesen, 1999). Increased illegal activity at night is a classic avoidance strategy, which has been observed in many previous fishery studies (e.g. Anderson, 1989; Crawford *et al.*, 2004; Ganapathiraju, 2012).

The adherence to spatial rules and regulations more generally was variable. Byelaw compliance was high at Boulmer, but low at Newton. The Boulmer byelaw is in place to protect local fishermen and their equipment when launching boats from the shore, whereas Newton is restricted for conservation reasons (UK Marine SACs Project, 2001a; NCAONB, 2009). Rules to protect structures (such as jetties and moorings) have also been observed to have higher compliance than those for conservation reasons in previous bait digging studies, perhaps due to the clarity of what is allowed and why, the associated shore user safety, and the additional deterrent of property damage litigation (Watson *et al.*, 2015). A conversation with one collector revealed that they only adhere to the Boulmer byelaw out of respect for the fishermen: "some people ignore the rules here, but I respect the fishermen and what they do too much to interfere around the boats" (questionnaire respondent, personal communication, 2015), suggesting conservation would be a lower driver of adherence for some individuals. Enforcement at both collection sites is low when both day and night are considered. NIFCA officers patrol the shores occasionally, and were observed at Boulmer a couple of times during daytime observations, whereas Newton has less

official enforcement from National Trust rangers or wardens on a day to day basis (but not night). Other unobserved restricted areas within Northumberland include Budle Bay, which is enforced by Natural England rangers from the Lindisfarne National Nature Reserve. Compliance is generally thought to be high in this area (Andrew Craggs, personal communication, 2015). However, again, night enforcement is lacking, and the occurrence or level of night time exploitation is unknown. Effective enforcement is critical to achieve a high level of compliance (Ceccherelli *et al.*, 2011; Cooke *et al.*, 2013; Watson *et al.*, 2015). Methods need to be face-to-face, as passive approaches such as signage, education, and codes of conduct alone have been ineffective in the past (Watson *et al.*, 2015). Increased enforcement of existing byelaws in the BNNC EMS is required to further reduce non-compliance, especially during the night.

2.4.2 Questionnaires – Collection Details

Participation of periwinkle collectors was low (11 individuals), a difficulty also encountered in other intertidal fisheries studies (e.g. Cummins *et al.*, 2002; Diogo *et al.*, 2016), as it can be difficult to openly study black economy industries due to the secrecy involved (McKay *et al.*, 1997; Cummins *et al.*, 2002). Lugworm collector participation was higher (66 individuals), and interviewers encountered mostly previous respondents or refusals towards the end of the observation period, suggesting that a representative sample of local collectors was achieved.

Questions which overlapped observable behaviours such as seasonality, collection method, and collection hotspots, agreed well with the patterns recorded from shore observations. This commonality validates the questionnaire data, an essential aspect of questionnaire design (Tashakkori and Teddlie, 2003), and suggests that generally, the questionnaire data can be considered reliable and reasonably representative.

There was a high degree of variability between respondents on aspects such as frequency of collection, duration of collection trips, and harvest quantity per trip. This is likely due to the high diversity of collectors for both activities – e.g. commercial vs recreational, competitive sport fishers vs casual leisure fishers, and full-time commercial collectors vs supplementary income commercial collectors (questionnaire respondents, personal communications, 2015). Individual collectors

have various motivations for harvesting lugworms and/or periwinkles, resulting in contrasting harvesting regimes and results (Fowler, 1999).

The average mass taken per collector per trip was 0.85 kg for lugworms, and 12.14 kg for periwinkles. Ragworm collection in the Solent appears to have higher catch rates than lugworms within Northumberland, with collectors harvesting an average 1.4 kg per hour at popular sites, equating to over 4 kg per tide (Watson *et al.*, 2017a). Bait digging bag limits elsewhere within the Solent (Pagham Harbour) are set at 0.5 kg per collector per visit (Watson *et al.*, 2015), resulting in lower harvest quantities per trip than those recorded in this study due to management. Periwinkle collectors clearly tend to harvest larger amounts each time than lugworm collectors. Although this collection activity appeared less popular in shore observations, the total harvest amounts are substantial due to these larger harvest quantities. This finding was also observed on shore, as periwinkle collectors often filled several large sacks (onion sacks (Crowley, 1975)), whereas lugworm collectors worked with much smaller capacity buckets (personal observation). Periwinkle fisheries in other parts of the world have substantially higher catch rates. In Tasmania, Australia, where harvesting is carried out by divers, a single days harvest (5 hours per day) can be 100-300 kg per fisher (Keane *et al.*, 2014).

Lugworm collectors showed strong preferences for sites with high quality lugworm stocks (large size and high density) over more practical considerations such as distance from home, ease of access, and parking availability. Conversations with collectors further confirmed this, as several stated that they travelled considerable distances to reach the best bait beds, with one stating that they travelled to Scotland (from South Northumberland) on occasion to maximise their harvest: "Edinburgh on a big tide is a good place to go, I collect there a couple of times a month" (questionnaire respondent, personal communications, 2015). Factors other than lugworm quality play a stronger role for some collectors than others, presumably linked with collection motivations. Site selection factors are important to consider, and can be used to map or model likely collector distribution (e.g. Bello-Pineda *et al.*, 2006; MMO, 2012; Villamagna *et al.*, 2014), which can be useful for spatial management (Jorgensen, 2011). Models trying to predict or represent lugworm collector 'habitat suitability' (Ortigosa *et al.*, 2000) (i.e. site selection) must consider

lugworm quality as a very influential factor (see Chapter 5 for an example model using lugworm quality as a significant indicator of collection intensity or probability).

2.4.3 Questionnaires – Commercial vs Recreational

There was a definite presence of commercial collectors within both fisheries. Commercial collectors seemed to be proportionally higher within the periwinkle industry (45% of respondents, compared to 14% lugworm respondents), likely due to the legitimacy of the activity and ease of sales direct to shellfish wholesalers (Cummins *et al.*, 2002). In comparison, commercial lugworm collection is not allowed, with no central buyer, which can explain the lower prevalence of commercial collectors within the fishery. Some commercial lugworm collectors appeared to be unaware that commercial collection was forbidden "as long as I don't dig in the no-digging zone I can collect and sell as much as I like" (questionnaire respondents, personal communications, 2015). Perhaps increased education of the rules and regulations is needed. One such commercial collector openly admitted to supplying local fishing tackle shops, which even contributed to collection expenses such as mileage: "the tackle shop gives me half my petrol money to get to Edinburgh on big tides" (questionnaire respondent, personal communications, 2015). Another popular sales avenue for commercial lugworm collection appears to be online using fishing related pages on popular social media sites (personal observation).

There were substantial differences in collection details (harvest quantities, time spent collecting per trip, and how often they collect) between recreational and suspected commercial collectors in both fisheries. Commercial collectors harvest more intensively – higher quantities, longer durations, and with greater frequency. Similar observations over collection durations have been inferred in previous studies (Watson *et al.*, 2015). The difficulties in proving commercial lugworm collection is well recognised (Watson *et al.*, 2015). Due to the use of the same collection methods, the two groups can be impossible to differentiate on the shore. With the added difficulty of personal bait storage systems increasing harvested worm longevity (e.g. Eguchi, 2001; Watson *et al.*, 2017a), it is difficult to ascertain a realistic harvesting quantity threshold to differentiate personal use from commercial sale. However, for management purposes it is critical to attempt to categorize the collection details of the two contrasting collector groups (commercial and

recreational) (Watson *et al.*, 2017a). This questionnaire data gives some useful discrimination between the two groups. With an average of 94 worms per recreational collector and 400 per commercial, a conservative estimate would be that an individual harvesting more than 200 worms per trip can be considered likely commercial. It is possible that some overlap will exist within this broad categorisation, such as recreational individuals whom fish very frequently (Armstrong *et al.*, 2013), and/or use long-term (weeks) storage solutions (Watson *et al.*, 2017a) exceeding the 200 worm threshold. A similar categorization can be done for periwinkle collection, with harvest quantities over 20 lbs broadly considered a commercial quantity.

Even with the lower proportion of commercial lugworm collectors compared to the periwinkle industry, the harvest amounts per individual are so much higher that overall, commercial collectors are estimated to harvest over 70% the total lugworms harvested within the BNNC EMS. For periwinkles this is extraordinarily high at over 95%. This highlights the importance of recognising the differences between commercial and recreational collection in future management plans, if commercial collection is having a disproportionately high impact.

2.4.4 Biomass Removal Estimates

Overall, a significantly higher biomass of periwinkles was harvested than lugworms (13.40 and 1.24 tonnes respectively). In terms of individuals, it is estimated that over 3 million periwinkles (estimated average periwinkle mass of 4 g based on quantities per kg from shellfish wholesalers (Yerseke, 2017)) and just over 200,000 lugworms are removed from BNNC EMS rocky and sandy shores each year.

It was previously estimated that 1,000 tonnes of bait worms are used in the UK each year (Fowler, 1999). If based on population size (National Office for Statistics, 2012), this would equate to approximately 5 tonnes within Northumberland, and if based on coastline length (The British Cartographic Society, 2008; NCAONB, 2009) 3.7 tonnes within the BNNC EMS. This would include other popular species such as ragworms (Fowler, 1999), and non-wild derived bait. An estimated lugworm harvest of 1.24 tonnes within the BNNC EMS seems relatively well matched with these previous UK wide estimates. However, more recent estimates of 3,400 tonnes of polychaetes

harvested annually from the UK (Watson *et al.*, 2017a) far exceed those observed in the BNNC EMS if averaged over the area or population. This suggests that baitworm collection within Northumberland may not be as significant as in other locations in the UK. For example, Watson *et al.* (2017a) estimated that 4.9 tonnes of ragworms are removed from Dell Quay in the Solent each year. Similarly, the annual harvest estimates for *D. neapolitana* in the Canal de Mira, Portugal, are vast, with 45 tonnes removed each year (Cunha *et al.*, 2005). These fisheries both translate to around 30 g harvested per m^2 (Watson *et al.*, 2017a). If the 1.24 tonnes of lugworms removed from the BNNC EMS was spread evenly over all sediment shores (29.04 km^2) the production value lies at around 43 mg per m^2. However, when Boulmer alone is considered (744.60 kg over 0.16 km^2) this figure rises to over 4 g per m^2 and close to 10 g per m^2 when only the legally harvestable area (high compliance) is considered. This higher production value is still three times lower than recorded for Dell Quay and Canal de Mira, however, is similar to those observed for ragworm collection at Fareham Creek (5 g per m^2) in the Solent, and the *G. dibranchiata* fishery (9 g per m^2) in Maine, USA (Watson *et al.*, 2017a). At the most intensively harvested shore, the production value per m^2 rivals those of other major bait worm fisheries both in the UK and abroad.

Annual periwinkle harvest estimates for Ireland and Scotland are both 4,000 tonnes (McKay *et al.*, 1997; Cummins *et al.*, 2002). Based on the McKay *et al.* (1997) estimate for Scotland, the BNNC EMS would have an estimated 25 tonnes when equated by coastline length (Scottish NCAONB, 2009; Government, 2011). This is around double the current estimate for the BNNC EMS (13.40 tonnes). However, Scotland is regarded as having a very large periwinkle industry, being the 6th most important fishery by mass, and 7th by value (McKay *et al.*, 1997), so this finding is not surprising. Additionally, Scotland's harvest may have reduced over the last 20 years since this estimate was made, as periwinkle collection was predicted to decrease over time by Cummins *et al.* (2002) based on an aging collector profile.

When the average estimated economic values of the BNNC EMS periwinkle and lugworm fisheries (£133,982 and £52,128 respectively) are compared to a value of £2.9 million for the Northumberland lobster fishery (Turner *et al.*, 2009), it is easy to see how the collection of lugworms and periwinkles can be overlooked in terms of

management, legislation, and research (McKay *et al.*, 1997). However, on a UK wide scale, the polychaete fishery is estimated to be worth £142 million per year, exceeding the lobster fishery by almost £40 million (MMO, 2013; Watson *et al.*, 2017a). The global polychaete fishery has recently been estimated at £5.9 billion (Watson *et al.*, 2017a). The acknowledgement of the high value of bait worm fisheries in recent years may lead to increased attention in the future regarding sustainability and management.

Many assumptions underpin these biomass removal estimates. Each component of the estimate (regular observations, broad scale observations, questionnaires) has associated uncertainty. Despite this, they are currently the only available biomass removal estimates for the BNNC EMS, and if used appropriately and conservatively, with acknowledgment of the weaknesses, they have the potential to help and support the creation of management plans. High and low scenarios of biomass and economic value are provided for this reason.

The contrasting harvest quantities of lugworms and periwinkles cannot be interpreted into impact levels, as the collection activities are very different, create different levels of associated disturbance, and the target species play very different roles within their ecosystems (e.g. Blake, 1979a; Janke, 1990; Townshend and O'Connor, 1993; Beukema, 1995; McKay *et al.*, 1997; Sharpe and Keough, 1998; Buschbaum, 2000; Berthelon *et al.*, 2004; Volkenborn and Reise, 2006; Volkenborn *et al.*, 2007b; Hidalgo *et al.*, 2008; Crossthwaite, 2012). Similarly, no assessments of sustainability can be attached to these estimates, as key aspects of fishery stock assessment remain unknown, for example: stock size, stock status, and spawning biomass (Smith *et al.*, 1993; Pitcher and Preikshot, 2001).

2.5 Conclusions

The need to assess intertidal fisheries has been acknowledged for many years (Olive, 1994; McKay *et al.*, 1997), yet they remain data poor, with inadequate information to support a harvest strategy or the implementation of control rules (Seafish, 2013). The lack of local and national fishery estimates creates challenges for managers (Watson *et al.*, 2017a). The common overlap with protected areas means that the lack of data makes the conservation of habitats, as well as fisheries management, difficult to implement with confidence (Watson *et al.*, 2017a).This chapter assesses the lugworm and periwinkle fisheries within the BNNC EMS, providing further evidence that intertidal fisheries can be significant, and should be assessed and considered in terms of management alongside other fisheries. The findings within this chapter supply localised fisheries data to managers, with the hope of informing future management plans under the requirement of the revised approach to commercial fisheries management in EMSs (MMO, 2014b).

This study has unravelled details of both the periwinkle and lugworm fishery within the BNNC EMS, despite the difficulties associated with such secretive and unreported industries (e.g. McKay *et al.*, 1997). Evidence of where, when, and at what intensity intertidal fisheries occur was previously lacking for both species (Moffat, 2015). The first quantitative (biomass and economic value) and spatial (broad scale collector distribution) assessments of the two fisheries are supplied, providing the best available evidence to managers. Future study could focus on finer scale collector distribution to inform 'within shore' management concepts, and continued monitoring is imperative for capturing changes over time.

Chapter 3 : Modelling the Suitability, Sensitivity, and Vulnerability of the BNNC EMS to Lugworm Collection

3.1 Introduction and Rational

The growth of marine activities over time, including fishing, has led to the amplification and diversification of human pressures on the marine environment (DEFRA, 2015), resulting in increasingly complex uses of marine space, and necessitating marine habitat and species protection worldwide (Douvere and Ehler, 2007). Marine managers aim to ensure sustainability, whilst minimising conflicts over resources and space (Jennings and Lee, 2012). Marine spatial planning (MSP), an emerging place-based management method stemming from the drive towards ecosystem-based management (Crowder and Norse, 2008), is implemented to help meet such aims (Douvere and Ehler, 2007; Douvere, 2008; Qiu and Jones, 2013). MSP is an integrated planning framework informing on the spatial distribution of a variety of marine activities, supporting current and future uses of marine ecosystems, whilst maintaining valuable ecosystem services for future generations (Douvere, 2008).

Fisheries management has an inherent spatial dimension (Douvere, 2008) well suited to MSP, with the definition of fishing grounds an important aspect to consider (Jennings and Lee, 2012). The management of fisheries applies to the resource users as much as the resource itself, as such there is a strong case for understanding the spatial dynamics of fishers (Turner, 2010), with the patchiness of fishing activities being an important consideration in the design of spatial marine management plans (Stelzenmüller *et al.*, 2008). Recent years have seen an increased focus on the ability of spatial management methods, for example marine reserves, to benefit fisheries (e.g. Gell and Roberts, 2003; Halpern, 2003; Sweeting and Polunin, 2005; Green *et al.*, 2014a; Lester *et al.*, 2017). Increased understanding of the distribution of fishers has the potential to further improve spatial management success, allowing for example: the prioritisation of protecting areas with lower fishing levels, the closure of areas with high fishing pressure for stock protection (Stelzenmüller *et al.*, 2008), the design of marine reserve networks (DEFRA, 2006), the identification of areas of economic importance to the fishing industry (Valcic, 2009), the assessment of fishery impacts, and the evaluation of resource management options (e.g. Pet-Soede *et al.*, 2001; Turner *et al.*, 2015) such as predicting responses of fishers to management (Valcic, 2009).

Within the field of marine fisheries management, large-scale and high-catch fisheries have received the most attention historically, with small-scale fisheries often lost in the market-based push towards sustainability (Jacquet and Pauly, 2008). There has been increased attention on small-scale fisheries in recent years (Berkes, 2003), but significant knowledge gaps remain. High resolution spatially accurate data are required to inform spatial management decisions (Eastwood *et al.*, 2007; Halpern *et al.*, 2012a; DEFRA, 2015), and the lack of such data in many small-scale fisheries raises both socio-economic and scientific concerns about the foundations of current spatial management decisions (Campbell *et al.*, 2014). Within the Berwickshire and North Northumberland Coast European Marine Site (BNNC EMS), information on the distribution of small-scale, especially intertidal fisheries, is lacking. Multiple fisheries within the BNNC EMS have been identified by Natural England and NIFCA, under requirements from Defra's revised approach (aka Fishing in MPAs project), as requiring further study to assess the impacts and inform management plans (MMO, 2014b). Lugworm collection from sandflats and mudflats has been identified as one area where data are lacking. Anecdotally, lugworms are collected widely throughout the BNNC EMS, however, there is currently no data available on the distribution of collectors, and it is unknown if the fishery is damaging the interest features of the conservation designations (Berwick and North Northumberland SAC and various SPA supporting habitats in the area) at current harvesting levels. To analyse the potential conflict between the nature conservation targets and the lugworm fishery, more data are required at the appropriate spatial scales (Pedersen *et al.*, 2009).

Common methodologies for mapping fishers distribution, or fishing pressure for inshore and offshore fisheries include the utilisation of existing spatial data in the form of fishery logbooks, plotters, enforcement and patrol surveys, or vessel tracking (Jennings and Lee, 2012). Vessel monitoring systems are considered a valuable data source for assessing fisheries spatially (Pedersen *et al.*, 2009). Within European seas, larger fishing vessels must operate a vessel monitoring system which transmits detailed information on the vessel location via satellites (O'Shea and Thompson, 2006). However, for smaller vessels without these systems (Turner, 2010), and intertidal fisheries not utilising vessels (such as collecting bait worms), this data is not available. Other Northumberland fisheries (e.g. lobster) have been mapped using Northumberland Inshore Fisheries Conservation Authority (NIFCA)

patrol sighting data in recent years (Turner, 2010; Turner *et al.*, 2015). However, this data does not exist for intertidal activities currently. NIFCA as part of the Fishing in MPAs project (MMO, 2014b) have been recording various activities along the coast and some lugworm collection data (anonymised) land-based patrols are becoming increasingly available but in low density and patchy distribution. With continued recordings, and an extended range, it is possible that NIFCA sightings data may be utilised in the future for mapping lugworm collection, using similar methods to those of Stephenson *et al.* (2017) for pot-fishing in Northumberland, accounting for patrol effort bias. However, until intertidal patrol sighting data are increased spatially and temporally, an alternate approach is required to map the lugworm fishery for which there is a lack of existing spatial data.

Spatial modelling techniques provide a cost effective and practical means of informing management decisions when data are lacking (e.g Sala *et al.*, 2002; Gritti *et al.*, 2006; Adams-Hosking *et al.*, 2011; Molloy, 2013). Fishing grounds or fisher distribution reflects choices by fishers, based on various factors such as: costs, past catch rates, agreements between fishers, hazard avoidance, and regulations (e.g. Gillis *et al.*, 1993; Babcock and Pikitch, 2000; Rijnsdorp *et al.*, 2000; Poos and Rijnsdorp, 2007). Similarly, coastal recreation distribution can be dependent on: ease of access, environmental quality, safety, and travel distance (Paudel *et al.*, 2011). Such choices or preferences can be used in models to predict where fishing is likely to occur for a particular fishing method. This is a land-use suitability model (Malczewski, 2004), an adapted habitat suitability model (Ortigosa *et al.*, 2000), which is used to predict areas of human activity based on environmental variables. Spatial models have been used to successfully map fishing and recreational activities in previous studies (e.g. Bello-Pineda *et al.*, 2006; MMO, 2012; Villamagna *et al.*, 2014; McIntyre *et al.*, 2015), and there is potential for lugworm fisheries to be modelled in similar ways, overcoming the current data gaps.

To fully inform potential management, it is important to relate the fishery distribution to the protected features or the sensitivity of the study site, identifying conflicts between the fishery and conservation aims (Young *et al.*, 2005). Describing the spatial distribution of fishing pressure alone can be useful for high-level management, but an understanding of the sensitivity of the targeted habitats makes the findings more meaningful at a local level (Bremner *et al.*, 2005; Stelzenmüller *et*

al., 2008; Tyler-Walters *et al.*, 2009). Sensitivity is a combined measure of how intolerant a species or habitat is to damage, and how long the subsequent recovery takes (MarLIN, 2010). A high fishing pressure does not result in a large impact if the habitat is not sensitive to that particular activity (Stelzenmüller *et al.*, 2008). This habitat sensitivity can also be modelled spatially. This has been done extensively for oil pollution using the Environmental Sensitivity Index (ESI), with the aim of prioritising clean-up efforts onto the most sensitive areas of the coast (Jensen *et al.*, 1998). For example, areas containing endangered species or high biodiversity are classed as more sensitive to oil spills (IPIECA *et al.*, 2012). Broad scale habitat sensitivity to fisheries has been modelled and mapped for inshore areas of both Ireland (Roberts *et al.*, 2010) and the Welsh part of the Irish Sea (Eno *et al.*, 2013) to inform site-specific fishery management plans. Modelling the sensitivity of the intertidal area of Northumberland to lugworm collection activities at a finer scale is possible, revealing the most sensitive areas of the coast to managers.

Sensitivity maps alone are not fully informative for marine management and planning (Roberts *et al.*, 2010), however when combined with details of fishing pressure distribution, they can demonstrate vulnerability of the habitat to the fishing activity. Vulnerability assessment is an increasingly popular method in various sectors, and provides a better understanding of interactions and threats, as a basis for targeted management strategies (Mamauag *et al.*, 2013). Vulnerability is a measure that combines information on sensitivity and exposure to an impact, for example, a habitat only becomes vulnerable when is it both sensitive to the activity and the activity is likely to occur there (Zacharias and Gregr, 2005; Roberts *et al.*, 2010). A vulnerability model can be produced by combining measures of suitability and sensitivity, which can be used by managers to target protection methods to the most vulnerable locations. Vulnerability assessments combining sensitivity and exposure level are used for pressure assessments for OSPAR sites (Roberts *et al.*, 2010), and similar theory can be applied to any combination of stressors and ecological features (Zacharias and Gregr, 2005). If both suitability and sensitivity are modelled for lugworm collection within the BNNC EMS, measures of vulnerability can be ascertained for each location, ultimately identifying specific areas for conservation purposes (Zacharias and Gregr, 2005).

Lugworm collection suitability and sensitivity have multiple factors within them, making it necessary to carry out multivariate analysis (Calenge, 2006), often termed multi-criteria evaluation (MCE) (Store and Kangas, 2001). In a management context this is also referred to as multi-criteria decision-making, were weights of preference are used to make better decisions, often using data layering processes to combine various criteria (Malczewski, 2004). Within this layering process there are two different methods commonly used, the Boolean overlay and weighted linear combination (WLC). Boolean overlay layers with 'and' and 'or' operations, whilst the WLC method standardises the suitability maps, assigning weights of importance to the various criterions (Malczewski, 2004), allowing for more complex relationships and a higher degree of detail to be included. There is no single accepted method for deciding criterion weights within WLC models. When empirical data is scarce, models can be built from the best available knowledge at the time, including patterns from previous studies (e.g. literature review), and expert knowledge (knowledge gained through training, education, or experience (Kuhnert *et al.*, 2010b), e.g. ecologists and fishermen) (Jorgensen, 2011). Using expert knowledge in fields where there is little published data is a cost-effective way to make more confident predictions (Martin *et al.*, 2005). Many studies have successfully incorporated expert knowledge into various types of ecological models (e.g. Store and Kangas, 2001; Martin *et al.*, 2005; Murray *et al.*, 2009, etc.). Regardless of the evidence source, it can be used and included in management decision making if it is judged to be relevant and trustworthy (Barends *et al.*, 2014). Where empirical data are lacking for lugworm fisheries within the BNNC EMS, expert knowledge appears to be a useful tool to inform spatial models.

This chapter aims to predict and describe the spatial patterns of lugworm fishing pressure within the BNNC EMS, relate the observed patterns to measures of sensitivity, and ultimately map the vulnerability of the study area to lugworm collection. Three spatial models are produced (suitability, sensitivity, and vulnerability), utilising data collected in the field (target species distribution), from literature review (impacts and sensitivities), and from expert knowledge (prioritising model criteria). Land-use suitability for lugworm collection is modelled using collector site preferences (e.g. target species quality, travel distance, etc.), sensitivity is modelled using recognised impacts associated with bait digging (e.g. especially

sensitive species or habitats), and finally both measures are combined to infer the environmental vulnerability. It is hoped that the models can be used to inform management plans for intertidal fisheries.

3.2 Methods

Lugworm collection suitability, habitat sensitivity to lugworm collection, and ultimately environmental vulnerability to collection are modelled spatially for all sediment shores falling within the BNNC EMS boundaries, using a WLC method in ESRI Arc GIS 10.4 software.

3.2.1 Model Requirements and Design

The literature on lugworm collection, site selection/preferences, and habitat and species sensitivities were first reviewed to identify key model criteria, along with author knowledge gained from encounters with collectors during questionnaires (Chapter 2), and key informant advice. The selected criteria for the suitability and sensitivity models, and the rationale and evidence base for each, can be seen in Table 3:1 and Table 3:2 respectively.

Based on a review of the spatial modelling literature, and the criteria identified, multi-criteria evaluation (MCE) was required (Store and Kangas, 2001; Malczewski, 2004). The Weighted Linear Combination (WLC) modelling method was selected for use in this study because of the standardisation of the output maps, due to weightings which allow multiple criteria to be combined effectively (Malczewski, 2004).

Table 3:1: Suitability model criteria, rationale, and evidence sources.

Criterion	Rationale	Evidence Source
Lugworm Abundance	Anglers generally target shores with a higher abundance of worms, as it makes the collection easier/more efficient.	Key Informant, Questionnaires
Lugworm Size	Anglers prefer larger worms and preferentially target them, therefore they would preferentially target shores which have larger worms available.	Fowler (1999), Key Informant
Black Lugworm Presence	Black lugworms are often a preferred choice for anglers. Additionally, bait pumps can only target black lug, so bait pumpers would only target shores with black lugworm present. Commercial collectors also get a higher price for black lugs.	Personal observations, Key Informant, Fowler (1999)
Sediment Type	Muddy sand appears to be the preferred sediment type by lugworm collectors. Mud is very difficult to work in, and sand is difficult to maintain trenches.	Key Informant, Personal observations
Distance to Parking	Access to parking is important for collectors, as they carry relatively heavy equipment and the lugworms in buckets.	Personal observations, Questionnaire
Distance to Home	Some commercial collectors are willing to travel long distances to harvest the best bait beds, however recreational anglers collecting for themselves are less likely want to travel too far since there is no financial gain to make it worth the extra distance. It is unlikely that they would travel further than a closer very suitable shore. Travel distance has been seen to influence beach choice for recreation.	Questionnaire, , Paudel *et al.* (2011)
Regulations	Areas without bait collection regulations are more suitable.	Personal observations, Key Informant

Table 3:2: Sensitivity model criteria, rationale, and evidence sources.

Criterion	Rationale	Evidence Source
Bird Importance	Bait digging is known to disturb birds from feeding on sandy shores. Shores which are important to birds may be the most sensitive to bird disturbance from bait digging.	Masero *et al.* (2008), Fowler (1999), Evans and Clark (1993)
Eelgrass Presence	Eelgrass species are sensitive to sediment disturbance, as uprooting damages it and it recovers very slowly. Therefore shores containing eelgrass beds are more sensitive to bait collection than those with none. It is also a very rare and important habitat.	Cabaço *et al.* (2005), Roberts *et al.* (2010), Mieszkowska (2010)
Sediment Type	Mud is more sensitive to physical disturbance since it is naturally more stable than sand and has less natural disturbance and movement. Since mud is more stable it also contains longer lived species which tend to recover more slowly. Therefore muddy shores are generally more sensitive than sandy shores to bait digging disturbance.	UK Marine SACs Project (2001a), MacDonald *et al.* (1996), Ferns *et al.* (2000), Roberts *et al.* (2010)
Lugworm Abundance	If the target species is already at a low abundance due to less suitable habitat or other environmental factors, that population will be more sensitive to exploitation, due to lower and slower recoverability.	Cunningham (2014), Cryer *et al.* (1987), Blake (1979a)
Lugworm Size	Larger lugworms have a higher reproductive output, and so better recoverability.	Pedersen *et al.* (2009), Watson *et al.* (1998)
Shore Isolation	Isolated shores such as pocket beaches surrounded by vast rocky shores would likely have poor recoverability. There would be no availability of close-by adult populations to migrate into disturbed patches and recruitment may be smaller. This is true for both the target species and the infaunal community as a whole.	Fowler (1999)

3.2.2 Data Collection

Spatial Data

Much of the spatial data required to inform the models were freely available from a variety of sources displayed in Table 3:3.

Table 3:3: Data Requirements to populate the models and the data sources used or identified.

Criterion	Data Source
Sediment Type	Biotope data from EMOD.net and sediment type data from OS Maps (accessed viahttps://digimap.edina.ac.uk/). Accessed May 2016.
Distance to Parking	Remote sensing using Google Maps (https://www.google.co.uk/maps/) satellite imagery and author knowledge of local parking sites. Accessed May 2016.
Distance to Home	'Large urban areas' identified from OS maps (accessed via https://digimap.edina.ac.uk/). Accessed May 2016.
Regulations	Literature and documented byelaw maps. Accessed May 2016.
Bird Importance	SPA designation boundaries from Natural England correspondence. Accessed May 2016.
Eelgrass Presence	Seagrass data from Environment Agency surveys. Accessed September 2016.
Shore Isolation	Remote sensing using Google Maps (https://www.google.co.uk/maps/) satellite imagery. Accessed May 2016.
Lugworm Abundance	No data available – field collection needed
Lugworm Size	No data available – field collection needed
Black Lugworm Presence	No data available – field collection needed

Most criteria were directly measurable or discrete, for example Regulations (there is either regulations in place, or not), and Distance to Parking (easily measured in spatial analysis software from satellite imagery). However, Bird Importance is not so easily defined. Bird sensitivity to disturbances such as bait collection is species specific (Davidson and Rothwell, 1993) (see Chapter 1, Table 1.2 for more detail on individual species potential impacts), and as such, species specific shore use data would be preferred for model accuracy. WeBS was considered as a data source, but proved unsuitable at the scale of the EMS as a whole. Therefore, SPA designation was chosen as a proxy measure for bird importance, assuming that areas chosen for bird protection would be highly sensitive to activities which cause bird disturbance, despite the lack of species specific impact data available.

Data on lugworm species presence, density, and body size were not available, so field data collection was required. Data collection points were generated using GIS to evenly distribute sampling across the EMS. Fishnet grids were randomly laid over the sediment shores on OS maps. The GPS of centroids of each grid square were

recorded for visitation in the field. Grids of 100m repetition were used for the majority of the study area, whilst 300m was used for more extensive areas of sediment (Budle Bay, Fenham Flats, and Goswick Sands (Figure 3:1) where a finer resolution was not appropriate for sampling (due to time and resource constraints with a substantially higher number of sample points at a finer resolution). Within these larger areas, not all identified coordinates could be visited in the field due to time and safety constraints. Areas that were sampled can be seen in Figure 3:1. All other shores within the BNNC EMS were sampled in their entirety at the 100m resolution.
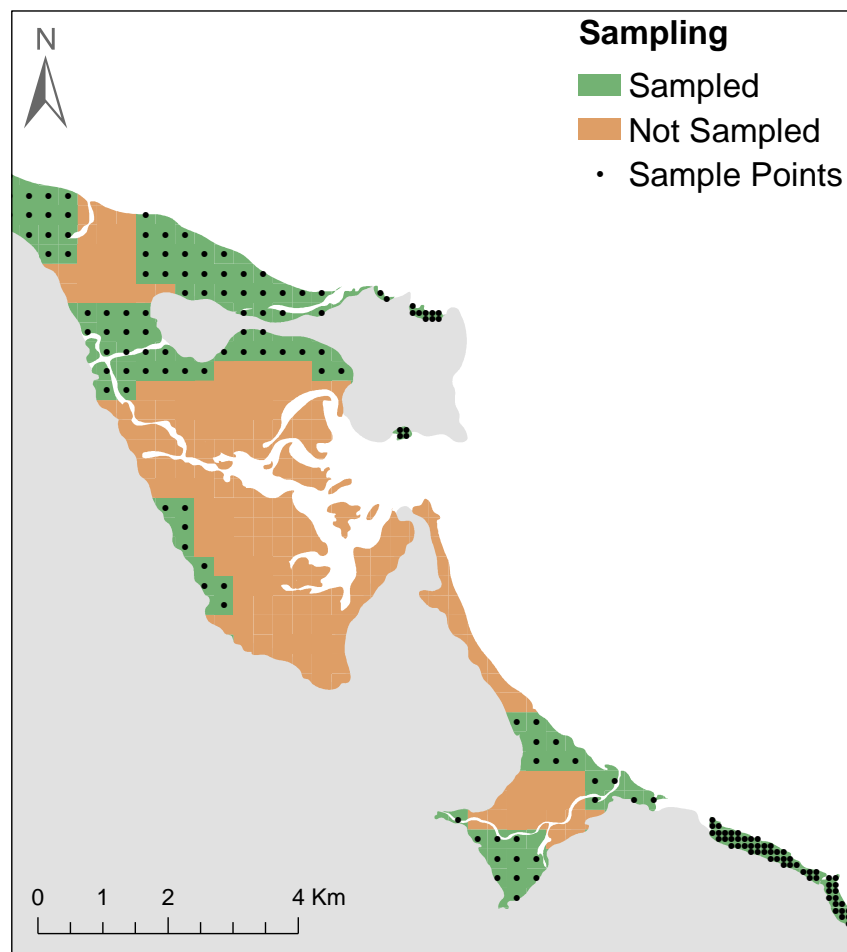
Figure 3:1: Sampled areas of Budle Bay, Fenham Flats, and Goswick Sands, where sampling was not able to cover the entire sediment area. Density of sample points corresponds to either 100m or 300m sampling resolution.

At each sample point the lugworm species were identified, and density and body size were recorded. Four replicate quadrats (1m^2) were randomly placed within 5 meters of the GPS point, to obtain averages for each grid square. Species identity was inferred from the faecal cast characteristics (Cadman and Nelson-Smith, 1993).

Lugworm density was recorded by counting the number of casts within the quadrat. Cast strand diameter was used as a proxy for lugworm size (Retraubun *et al.*, 1996b), which was recorded with callipers to the nearest millimetre for ten randomly chosen casts per quadrat. At coordinates where lugworms were present in low abundances but not recorded within quadrats, a mean density of 0.1 per m^2 was assigned within the model, to give the most representative lugworm distribution/density maps possible (no grid squares recorded as containing no lugworms when they were present in low density).

Non-Spatial Data

The weightings for model criteria were determined by interviews with experts. Experts were sought from a variety of backgrounds (conservationists, land managers, academics, and angling) to reduce bias. The consulting experts or their organisations can be seen in Table 3:4.

Table 3:4: Experts/Organisations consulted for opinions on the importance of each criteria within each model

Authority/Employer	Suitability	Sensitivity
Author - Personal Observations and Key Informants	✓	✓
Natural England	✓	✓
NIFCA	✓	✓
Northumberland Wildlife Trust		✓
Angling Trust	✓	
Academic – Newcastle University		✓

Expert interviews were conducted face to face where possible, with participants asked to rank the model criteria in order of importance. The resulting rankings were subsequently averaged to aggregate the multiple responses (Kuhnert *et al.*, 2010b) and converted into model weights (highest weight = most important), which can be seen in Table 3:5 and Table 3:6. An adapted Delphi approach was used for elicitation (Kuhnert *et al.*, 2010b), ensuring all experts were satisfied with the

combined results, and allowing for adequate feedback prior to incorporation within the models.

Table 3:5: Mean and resulting criterion ranked scores for the suitability of lugworm collection (averaged from 4 experts), with corresponding model weights

Criterion	Lugworm Abundance	Lugworm Size	Black Lugworm Presence	Sediment Type	Distance to Parking	Distance to Home	Regulation
Average Ranking	1.75	2.5	2.5	4	5.25	6	6
Resulting Ranking	1	2	2	3	4	5	5
Weighting	5	4	4	3	2	1	1

Table 3:6: Mean and resulting criterion ranked scores for the sensitivity of lugworm collection (averaged from 5 experts), with corresponding model weights

Criterion	Bird Importance	*Zostera* spp Presence	Sediment Type	Lugworm Abundance	Lugworm Size	Shore Isolation
Average Ranking	2.6	2.8	2.2	4	4.6	4.8
Resulting Ranking	2	3	1	4	5	6
Weighting	5	4	6	3	2	1

Weightings allow the most important factors to have more influence over the model outputs (Malczewski, 2004). Within the suitability model criteria, lugworm abundance was the selected as the most influential factor for collectors selecting where to harvest. This is due to bait diggers preferentially targeting shores with high numbers of lugworm, with the aim of exerting less effort (amount of sediment overturned) for the same return (number of worms harvested) (expert interviews, personal communications, 2016). Bait digging regulations were assigned the lowest weighting by experts for the suitability model because collection is known to occur illegally despite regulations, and although they may deter some collectors, it does not make the site unsuitable overall (expert interviews, personal communication, 2016). Non-compliance with these regulations has been observed within the BNNC EMS (Chapter 2), and the UK more widely (Watson *et al.*, 2017a). The suitability model aims to map where collection likely occurs, rather than where it should or should not occur. Therefore, the model outputs may show areas with bait digging restrictions as highly suitable, despite the obvious unsuitability for managers.

For the sensitivity model, sediment type was considered the most influential factor by experts due to bait digging and pumping being better suited to certain sediment conditions (expert interviews, personal communications, 2016). Bird importance was also considered highly influential, as birds are a major classified feature of many of the conservation designations within the BNNC EMS, and are known to be sensitive to the disturbance associated with bait collection (Masero *et al.*, 2008; NCAONB, 2009; experts interviews, personal communications, 2016). Shore isolation was considered the least influential factor of environmental sensitivity, due to experts prioritising factors which directly affect sensitivity via the intolerance level of the habitat or species to disturbance, rather than the rate of recovery if impacts were significant (expert interviews, personal communications, 2016).

3.2.3 Model Building

ESRI Arc GIS 10.4 software was used to manipulate the spatial data and build the models.

Model Contents

Separate models were constructed for suitability and sensitivity, which were subsequently combined to provide a measure of vulnerability (Zacharias and Gregr, 2005; Roberts *et al.*, 2010), seen in Figure 3:2.

All data, existing and field collected, needed some manipulation and reclassification to create thematic data layers. The criteria were standardised using a scoring system for the sub-categories within (e.g. size classes of lugworm within the lugworm size criterion), which is required for MCE (comparable units) (Hossain and Das, 2010). A scale of 0-6 was used for both models, with a score of 6 signifying the most suitable or sensitive category. The numerical definitions of each sub-category with continuous data were calculated in Arc GIS using Jenks natural breaks optimization to cluster the data appropriately, minimising each classes average deviation from the class mean, and maximising that between classes (Jenks, 1967). The sub-categories within non-continuous data criteria were selected based on discrete, more descriptive classes, e.g. mud or sand. The class direction of scoring within categories was decided using expert opinion gained during the expert interviews for weightings, questionnaire responses and key informant conversations with bait

collectors (see Chapter 2 for more details), and literature research. The criteria, sub-categories, descriptions, scores, and justifications can all be seen in Table 3:7 and Table 3:8 for the suitability and sensitivity models respectively.
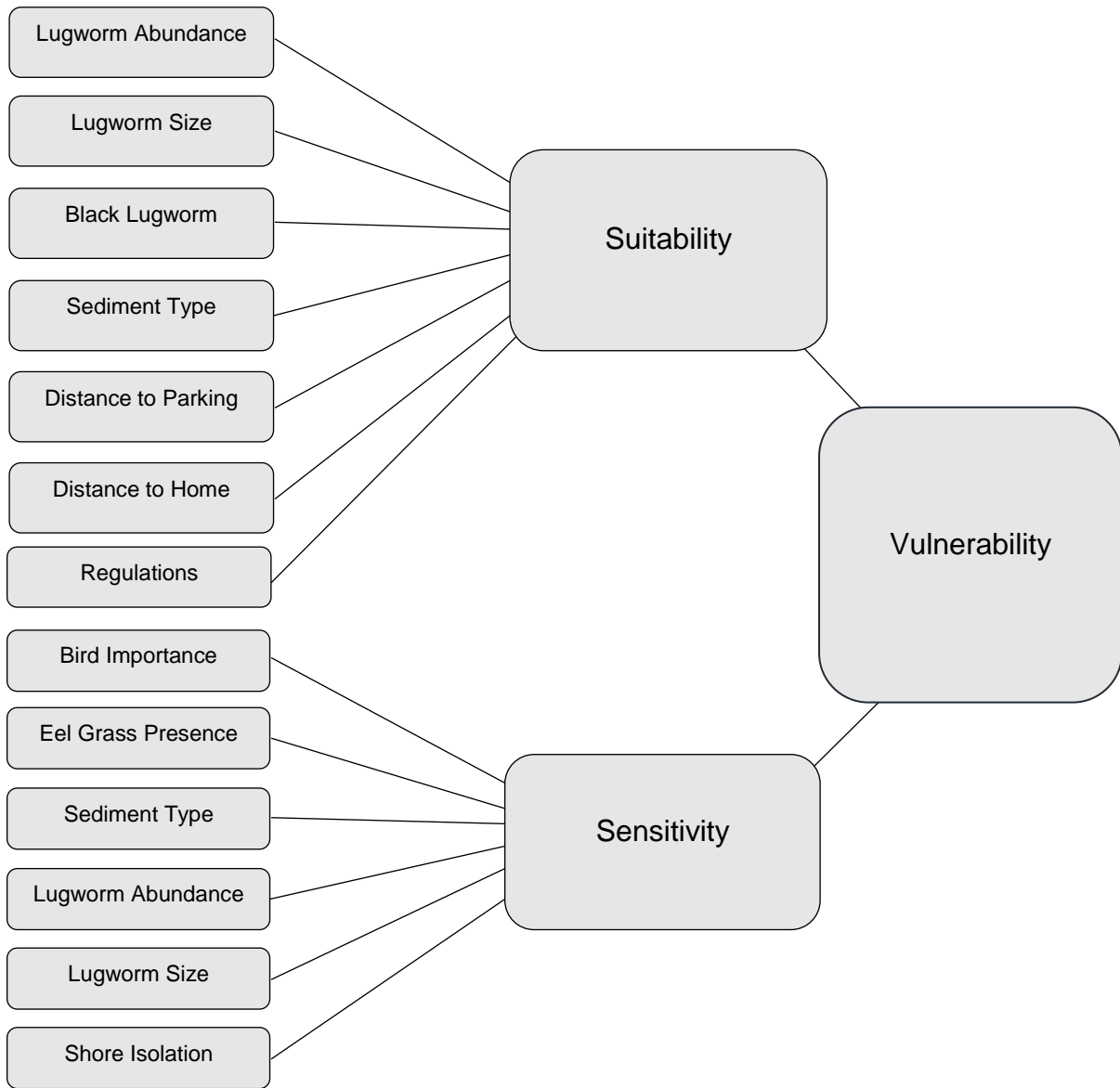
Figure 3:2: Conceptual model diagram. Far left are the model criteria and input data layers, middle is the two major model outputs, and far right is the final model output.

Table 3:7: Suitability model criteria scoring: sub-categories, definitions, scores, and justifications.

Criterion	Sub-categories	Definition	Score	Justification and Evidence
Lugworm Abundance	Absent Very Low Low Moderate High Very High Extremely High	0 0 – 9.75 per m^2 9.75 – 16.25 per m^2 16.25 – 22.75 per m^2 22.75 – 43.75 per m^2 43.75 – 81.75 per m^2 81.75 – 160 per m^2	0 1 2 3 4 5 6	The more lugworms present, the easier collection becomes (less effort per worm), and as such densely populated shores are most popular with collectors (Expert opinion interviews, personal communications, 2016; Collector questionnaires, personal communications, 2015). More worms = higher suitability. Groupings selected by Jenks breaks optimization from field data spread.
Lugworm Size	Absent Very Small Small Moderate Large Very Large Extremely Large	0 0 – 1.7 mm 1.7 – 2.3 mm 2.3 – 2.8 mm 2.8 – 3.2 mm 3.2 – 3.75 mm 3.75 – 4.6 mm	0 1 2 3 4 5 6	Anglers prefer larger worms (Fowler, 1999), as such, shores with the biggest worms present are favoured by collectors (Expert opinion interviews, personal communications, 2016; Collector questionnaires, personal communications, 2015). Bigger worms present = higher suitability. Groupings selected by Jenks breaks optimization from field data spread.
Black Lugworm Presence	Absent Present	N/A	0 6	Black lugworms are often preferred by anglers, commercial collectors are paid more for them, and bait pumpers can only target them (Fowler, 1999). Black lugworms present = highly suitable.
Sediment Type	Mud Sand Muddy sand or sandy mud	N/A	2 4 6	Muddy sand is the preferred sediment type targeted by collectors, with sand being less suitable due to the texture not maintaining trench structure when digging, and mud even less so due to the challenges of moving around and digging in sticky mud (Expert opinion interviews, personal communications, 2016; Collector questionnaires, personal communications, 2015).
Distance to Parking	Very Far Far Medium Close Very Close Extremely Close	1616 – 2344 m 1144 – 1616 m 773 – 1144 m 446 – 773 m 229 – 466 m 0 – 229 m	1 2 3 4 5 6	Parking proximity can be important for some collectors, especially older individuals. Close parking is more convenient when carrying digging forks and buckets full of seawater and lugworms (Collector questionnaires, personal communications, 2015). Closer parking = more suitable. Groupings selected by Jenks breaks optimization from distance data spread.
Distance to Home	Very Far Far Medium Close	18789 – 21611 m 16196 - 18789 m 13513 – 16196 m 9827 – 13513 m	1 2 3 4	Although some collectors are willing to travel extremely far to access to the best bait beds, many regular local anglers prefer to collect close to home for convenience and cost savings (Collector questionnaires, personal communications, 2015).

	Very Close	4267 – 9827 m	5	Closer to home = more suitable (if bait quality the same).
	Extremely Close	0 – 4267 m	6	
Regulations	Enforced	Holy Island and Budle Bay	2	Despite non-compliance of some individuals, overall, bait digging byelaws
	Not well enforced	Newton and Boulmer South	4	do deter most collectors, reducing the suitability for collection (Collector
	No Regulations	Rest of EMS	6	questionnaires, personal communications, 2015). More enforcement is a
				greater deterrence (Nielsen and Mathiesen, 1999), as such, Holy Island
				and Budle Bay are given the lowest suitability score since rangers patrol
				the areas regularly, compared to little enforcement at Newton and Boulmer.

Table 3:8: Sensitivity model criteria scoring: sub-categories, definitions, scores, and justifications.

Criterion	Sub-categories	Definition	Score	Justification and Evidence
Bird Importance	No bird designation	N/A	2	Birds can be negatively affected by bait digging (Masero *et al.*, 2008).
	Northumbria Coast SPA		4	Areas which are protected for birds (SPAs) contain sediment shores where bird disturbance may be particularly harmful due to higher bird abundance
	Lindisfarne SPA		6	or rare bird refuges (Expert opinion interviews, personal communications, 2016). Lindisfarne SPA has the most protected species designations (NCAONB, 2009), and as such was given the highest sensitivity score. Areas outside of SPAs are still sensitive, and as such are given a lower score of 2.
***Zostera* spp Presence**	Absent	N/A	0	Eelgrass is sensitive to physical disturbance from bait digging (Mieszkowska, 2010), and are protected by a no digging byelaw in the
	Present		6	BNNC EMS. Only areas containing Eelgrass are sensitive to eelgrass disturbance, and as such are given the highest score of 6, with all other areas not sensitive with a score of 0.
Sediment Type	Sand	N/A	2	Mud is most sensitive to digging disturbance due to the stable nature, and
	Muddy sand or sandy mud		4	presence of longer lived, slower recovering species. Sand is mobile in nature and recovers faster from disturbance, making it the least sensitive to
	Mud		6	bait digging (Ferns *et al.*, 2000; Roberts *et al.*, 2010).
Lugworm Abundance	Extremely High	81.75 – 160 per m^2	1	Larger/more dense populations are less sensitive to over exploitation, as
	Very High	43.75 – 81.75 per m^2	2	smaller proportions of the populations are harvested, and recovery will be
	High	22.75 – 43.75 per m^2	3	faster with more reproductive contributions (Cryer *et al.*, 1987;
	Moderate	16.25 – 22.75 per m^2	4	Cunningham, 2014).
	Low	9.75 – 16.25 per m^2	5	The more worms, the less likely the population can be overexploited, and
	Very Low	0 – 9.75 per m^2	6	the less sensitive they are.

	Absent	0	0	Areas with no lugworms are not sensitive at all (score of 0), since they will not be harvested.
Lugworm Size	Extremely Large	3.75 – 4.5 mm	1	Larger worms have a greater reproductive output with more eggs produced per individual (Watson *et al.*, 1998). Shores with larger individuals present will have a higher reproductive output, and therefore higher recoverability. Larger average worm size = less sensitive to overexploitation.
	Very Large	3.2 – 3.75 mm	2	
	Large	2.8 – 3.2 mm	3	
	Moderate	2.3 – 2.8 mm	4	
	Small	1.7 – 2.3 mm	5	
	Very Small	0 – 1.7 mm	6	
	Absent	0	0	
Shore Isolation	Not Isolated	>3000m^2, <500m apart	0	Small and isolated beaches have lower recoverability due to lower recruitment rates from adjacent shores (Fowler, 1999). Most sediment shores within the BNNC EMS are relatively close to each other, so a distance between shores of more than 500m was chosen to differentiate a few more isolated shores from the rest. Similarly, many of the shores are large, so an area of less than 3000m^2 was considered small based on the measurements from all shores. A shore which was both less than 3000m^2 and separated by more than 500m of rocky shore or cliffs was considered a 'pocket beach' and regarded as a small isolated shore with high sensitivity.
	Isolated	<3000m^2, >500m apart	6	

Constructing the Models

The shape file of the sediment shores within the study area was converted to a grid with 100m cell size. The geospatial data for each criterion were imported into replicated grids, retaining the same geographic extent and resolution as the base layer, standardising the individual criteria layers. The criteria grid layers were reclassified based on the suitability or sensitivity scores assigned from Table 3:7 and Table 3:8, totalling 3808 squares filled with the relevant scores per layer. Where no data were available for a particular grid square (e.g. no field data available either from no samples in an area or a lower resolution of 300m sampling on larger shores), interpolation methods were used to fill all squares, assuming conditions were similar in close proximity. Once all grid squares contained standardised scores, a requirement of MCE (Hossain and Das, 2010), each data layer was multiplied by the appropriate weightings in Table 3:5 and Table 3:6. Within each model (suitability and sensitivity), all criteria data layers were combined into a single layer with a summed total score. The combined suitability scores were further multiplied by 0 if no lugworms were present within the grid square (density recorded as 0 in the model), and 1 if lugworms were present, to control for areas without lugworms being categorised as suitable for collection due to other high scoring criteria. The suitability and sensitivity models were finally combined to produce the vulnerability model (product of both suitability and sensitivity scores). The final scores for suitability, sensitivity, and vulnerability were split equally into 6 groupings using Jenks natural breaks optimization (Jenks, 1967), which can be seen in Table 3:9. The GIS analytical steps can be seen in more detail in Figure 3:3.

Table 3:9: Final suitability, sensitivity, and vulnerability scores and classes.

Suitability Score	Suitability Class	Sensitivity Score	Sensitivity Class	Vulnerability Score	Vulnerability Class
0	Unsuitable	22-47	Very Low	0	Very Low
1-46	Low	48-62	Low	1-2940	Low
47-56	Moderate	63-77	Moderate	2941-3760	Moderate
57-67	High	78-90	High	3761-4623	High
68-84	Very High	91-102	Very High	4624-5504	Very High
85-106	Extremely High	103-118	Extremely High	5505-6474	Extremely High

Model Set-up

- Shapefile of study area imported (OS map at 1:50K)
- Polygons created for all sediment shores (tracing the OS map)
- Fishnet tool to create 100m and grid overlay
- Geoprocessing clip tool to join grids onto the shore polygons, and exported into new data layer
- GPS coordinates of grid centroids calculated (calculate geometry tool)
- Sample points named in attribute tables

Importing Data

- Exsisting spatial data imported and transformed to standardised coordinate system (BNG) - e.g. SPA shapefiles. Subsequently transferred into duplicated grid shapefile, so that each grid sqaure contained data on the criteria
- Shapefiles created for criteria which need manipulation of exsisitng data - e.g. distance from home
- Data maipulated spatially - e.g. near (analysis) tool used to find distance to parking or population centres from each grid square
- Shapefiles created for field collected data (e.g. lugworm density) and data imported from excel into each grid square via the attribute table.
- Data interpolated into unsampled squares using nearest neighbour tool
- Resulting in each criterion as a separate data layer, all with the same base grid within shore polygons - spatially standardised

Model Building

- Criteria split into relavant sub-categories using jenks natural breaks optimization tool for continuous data - e.g. distance to parking
- Scores assigned to each sub-category for each criteria within the attribute tables (scores of 0-6, see Tables 3.7 and 3.8).
- Weighted overlay tool uses rasta layers only, so manual overlay used to maintain detail and editability
- Each grid square score multiplied by the relevant weighting (Tables 3.5 and 3.6) using the field calculator within the attribute table for each criteria data layer
- All criteria data layers joined for each model (suitabilty and sensitivity) to give a single data layer for each model containing all criteria within the attribute table

Running the Models

- Within each combined model data layer, the final score for each grid square is calculated using the field calulator by summing the previously weighted scores for each criteria in the attribute table
- Final scores are split into classes of suitability and sensitivity using jenks natural breaks optimization (Table 3.9)
- Colour gradients are applied to the classes to display the final scores visually
- Finally, both models (suitability and sensitivity) are joined into a new data layer, with the scores of each multplied together (using field calculator) to give an overall score representing vulnerability, which were similarly split into classes (Jenks) and displayed

Figure 3:3: Methods of GIS analytic steps used in creating the models. Software = ESRI Arc GIS 10.4.

3.3 Results

3.3.1 Model Inputs – Suitability and Sensitivity to Lugworm Collection

Figure 3:4 and Figure 3:5 show the individual data layers created to populate the sensitivity and suitability models respectively. These maps display the most diverse section of the study area between Budle Bay and Beadnell only, as clear depiction at the appropriate resolution is not possible for the entire BNNC EMS due to the large size (NCAONB, 2009).
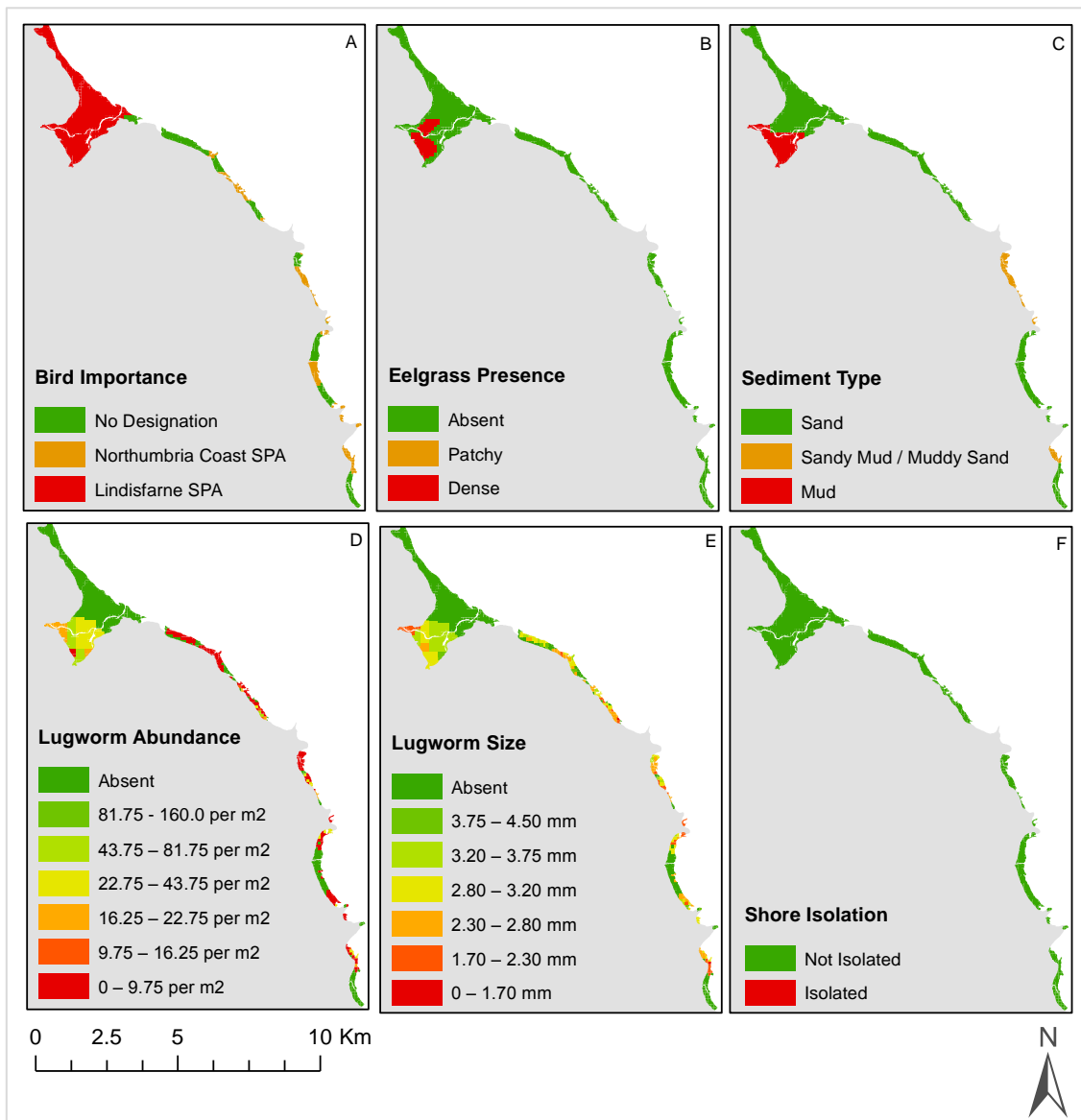
Figure 3:4: Input criteria data layers to populate the lugworm collection sensitivity model: a) Bird importance, b) Eelgrass Presence, c) Sediment type, d) Lugworm abundance, e) Lugworm size, f) Shore Isolation. Red is the highest sensitivity, green the lowest sensitivity. Displaying Budle Bay to Beadnell Bay. See Appendix B for aerial images of the aspect shown here.
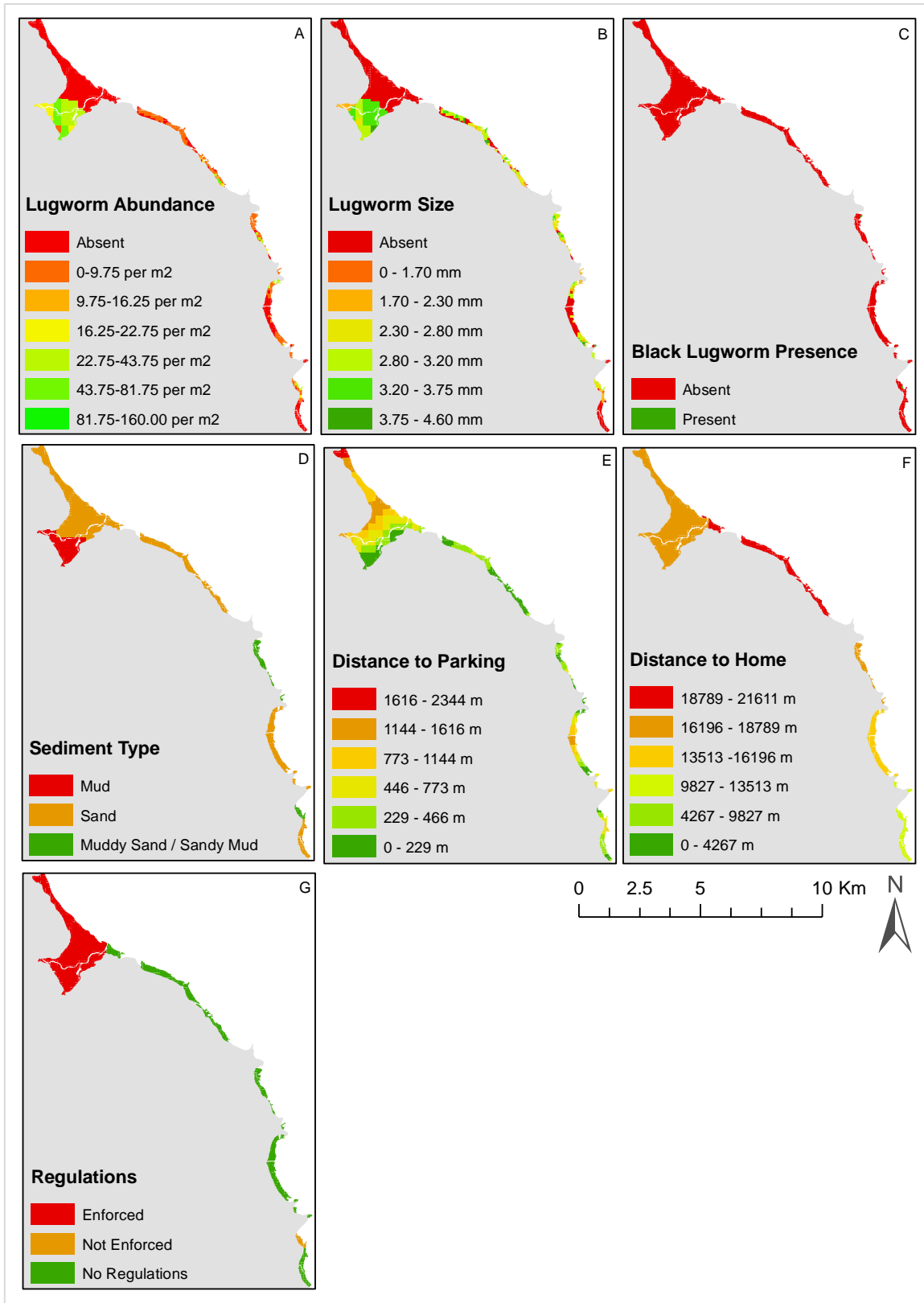
Figure 3:5: Input criteria data layers to populate the lugworm collection suitability model: a) Lugworm abundance, b) Lugworm size, c) Black lugworm presence, d) Sediment type, e) Distance to parking, f) Distance to home, g) Regulations. Red is the lowest suitability, green the highest suitability. Displaying Budle Bay to Beadnell Bay. See Appendix B for aerial images of the aspect shown here.

The field collected data maps (lugworm density, size, and species) are the first lugworm distribution and population maps available for the study area. The highest mean lugworm density was 156 lugworms per m^2, which was recorded at Holy Island on the northern coves. There was much variation in lugworm distribution and density both between and within shores Figure 3:6.
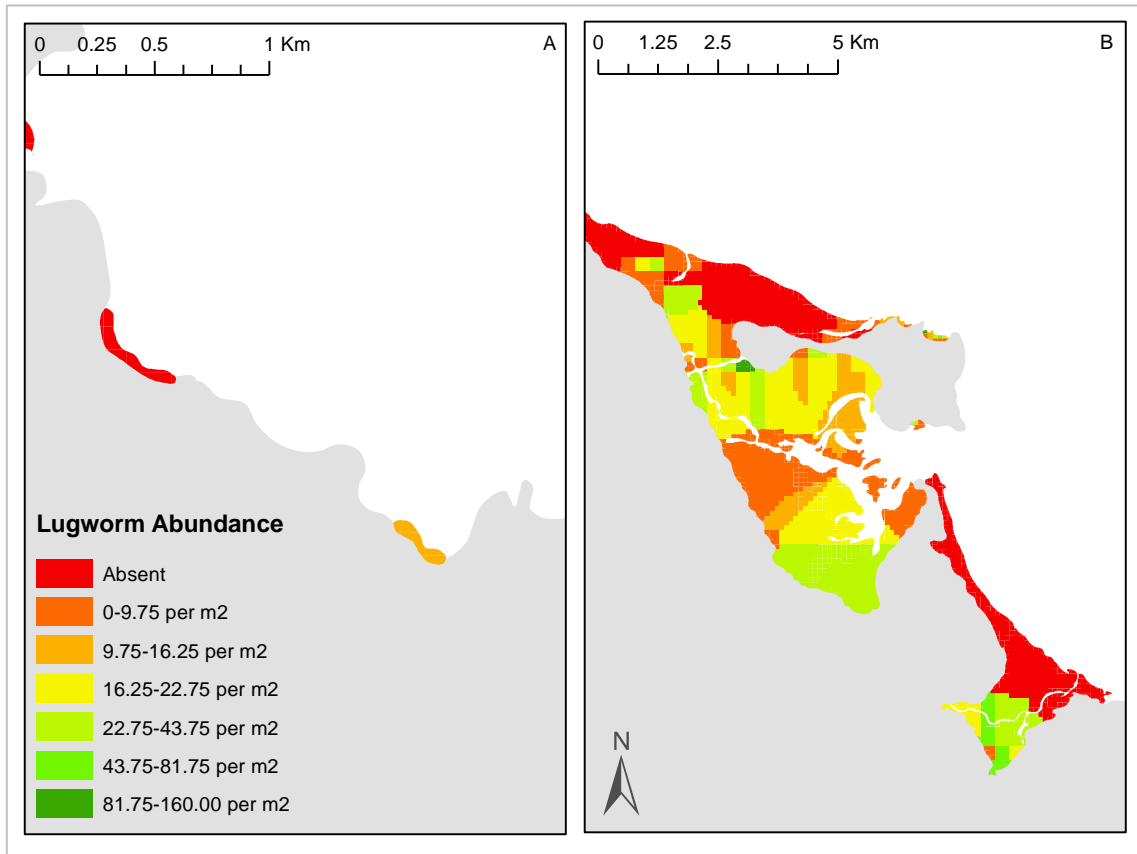
Figure 3:6: Lugworm abundance maps in more detail, displaying variation between shores (a – Killiedraught Bay and Coldingham Bay) and within shores (b – Holy Island, Fenham Flats, and Budle Bay). See Appendix B for aerial images of the aspects shown here.

3.3.2 Model Output – Suitability for Lugworm Collection

The suitability of sediment areas for lugworm harvesting were estimated for the whole of the BNNC EMS, acting as a predictor of lugworm collection activity. The more suitable an area, the more likely collection takes place there. The model output map (Figure 3:7 a) depicts the sediment shores as varying levels of suitability, ranging from unsuitable to extremely high suitability. The degree of suitability for lugworm harvesting varies widely both between and within shores. Only 1% of the sediment area was classified as having extremely high suitability, the locations of

which can be seen in more detail in Figure 3:7 (b,c, and d). These zones included parts of the sediment shores at Berwick, Newton, and Boulmer.

If the suitability model outputs are to be regarded as a valid predictor of lugworm collection activity, they need to be validated (Jorgensen, 2011). The most suitable areas (i.e. the most likely collected) were compared to actual collection activity previously recorded in Chapter 2 using shore observations. There is high similarity between the zones categorized as having extremely high suitability, and those with the highest recorded collection and biomass removal in Chapter 2. Both the between shore and within shore zones match well with observations of collector distribution and collection intensity, suggesting that the model successfully predicts the most suitable areas for collection, which does in turn translate into collection pressure.

Other suitable areas are likely collected at a lower intensity. High suitability and very high suitability areas identified by the model include: the far north of Foxton, the North of Boulmer, a small patch of Longhoughton, Newton central shore, Football Hole, far north and south of Beadnell Bay, patches of Seahouses and North Sunderland, small areas of Bamburgh, Budle Bay inland, Fenham Flats, Holy Island north shores, Berwick north shores, and Eyemouth. It is predicted that these areas are also targeted for lugworm collection to some degree (sometimes illegally).
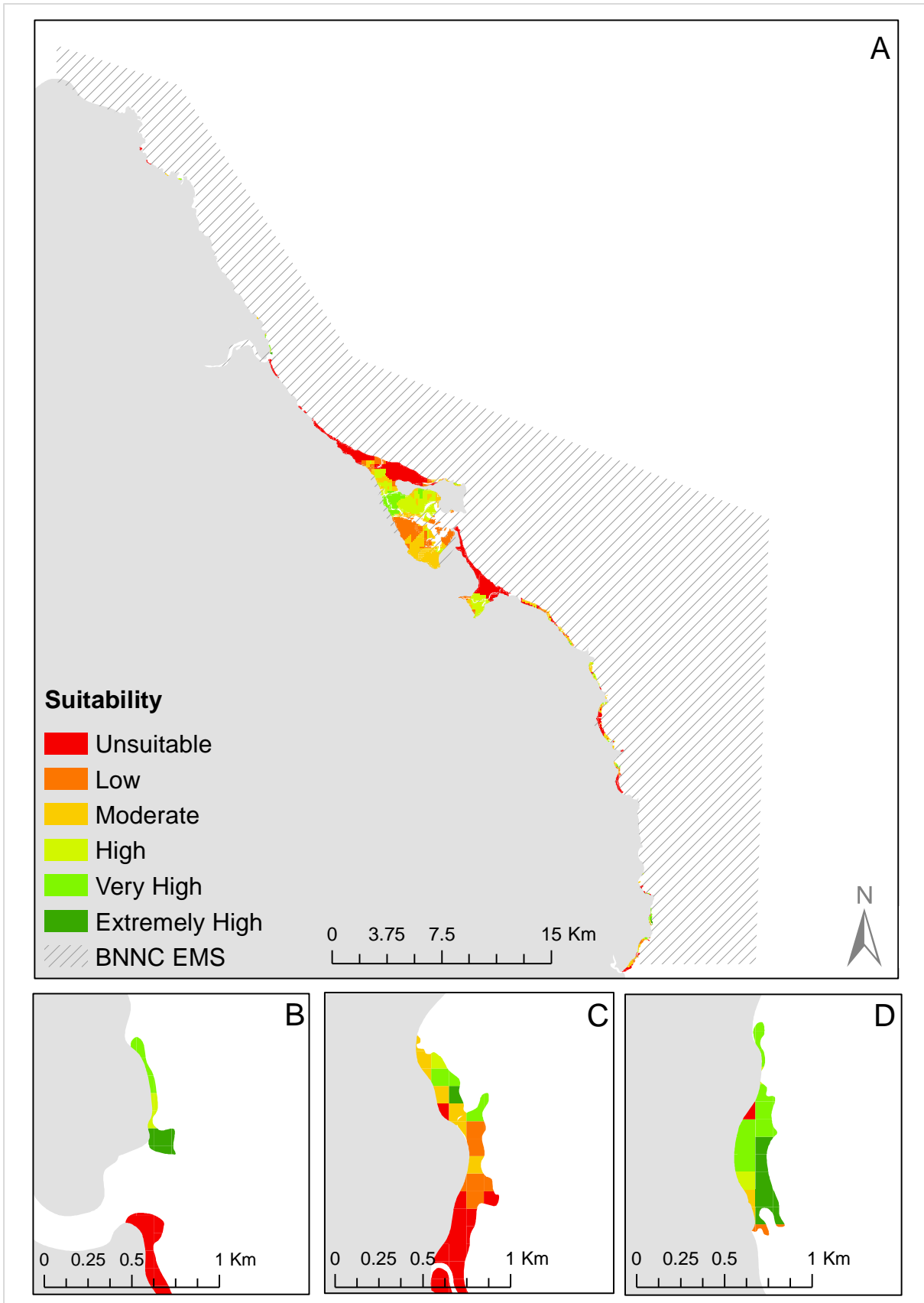
Figure 3:7: Suitability model output for the BNNC EMS (a), with the most suitable areas shown in greater detail: (b) Berwick, (c) Newton, (d) Boulmer. See Appendix B for aerial images of the aspects shown here.

3.3.3 Model Output – Sensitivity to Lugworm Collection

The sensitivity of sediment areas to lugworm harvesting were estimated for the whole of the BNNC EMS, acting as a predictor of the level of impacts associated with lugworm collection activity. The more sensitive an area, the more severe the associated impacts are likely to be there. The model output map Figure 3:8 (a) depicts the sediment shores as varying levels of sensitivity, ranging from very low sensitivity to extremely high sensitivity.

Just under 14% of the sediment area was classified as having extremely high sensitivity, the locations of which can be seen in more detail in Figure 3:8 (b and c). These zones included parts of the sediment at Fenham Flats and Budle Bay, suggesting that the habitats and species within parts of the Lindisfarne National Nature Reserve would be the most sensitive to lugworm collection – i.e. the largest and longest lasting negative impacts (Roberts et al., 2010(MarLIN, 2010)). These areas are generally muddy, important to birds (SPA area), and form sea grass habitats.

Other areas classified as having high or very high sensitivity include further areas of Budle Bay and Fenham Flats, as well as Holy Island southern and northern shores. The remaining areas of sediment within the BNNC EMS have lower measures of sensitivity, but it is important to acknowledge that damage/impacts on designated and classified features from bait digging is still possible in all locations.

Unlike the suitability model, the outputs from the sensitivity model cannot be validated externally, due to no similar but independent data cohort to make comparisons against (Salciccioli *et al.*, 2016). Sensitivity is not measurable in the field, and therefore this model could not be validated using experimental data (Trucano *et al.*, 2006). It is important to note that these areas have been previously identified as being sensitive to bait digging (and other activities) and there is extant management in place (e.g. NNR byelaws and NIFCA byelaws) to protect the designated and classified features.
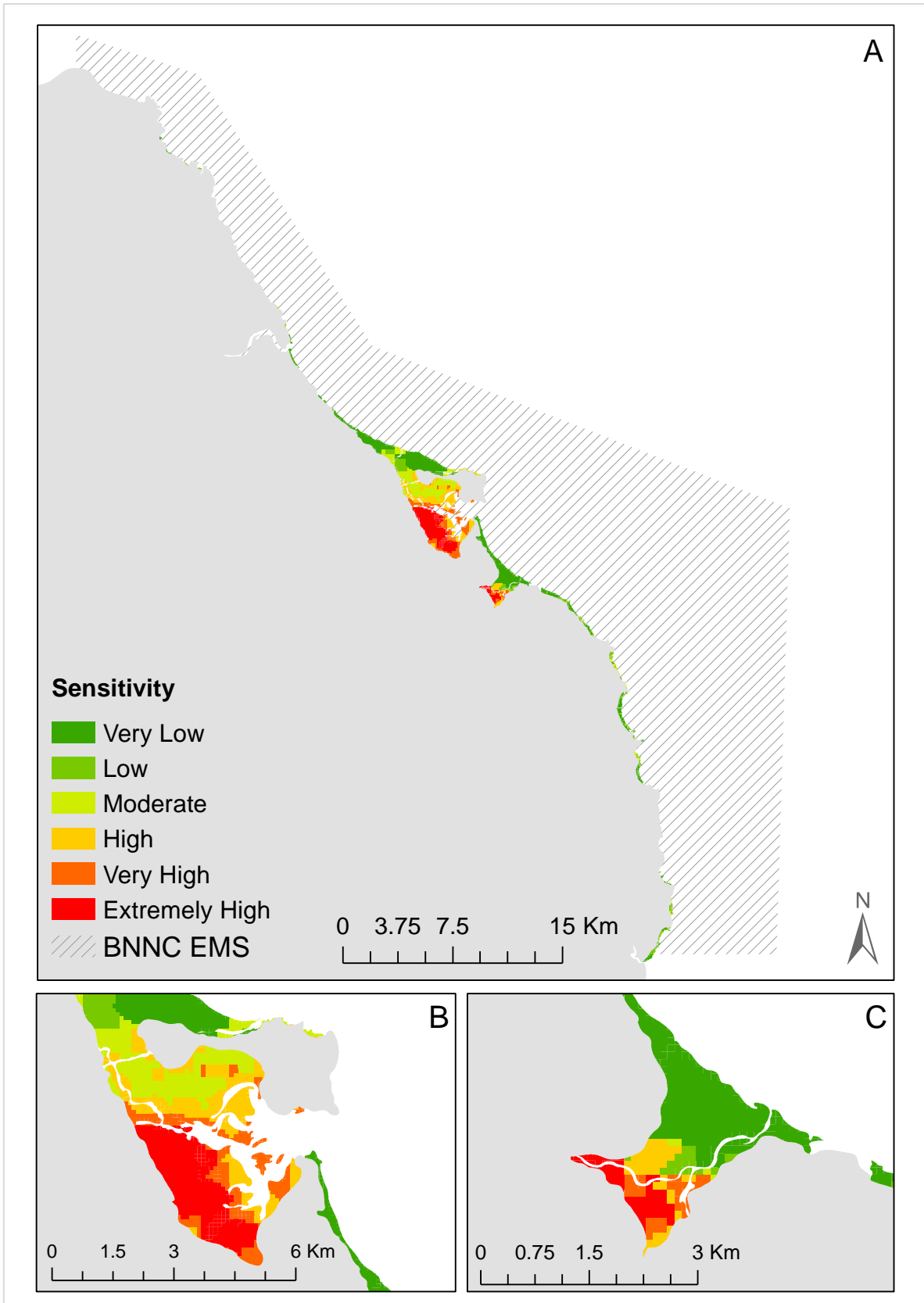
Figure 3:8: Sensitivity model output for the BNNC EMS (a), with the most sensitive areas shown in greater detail: (b) Fenham Flats, (c) Budle Bay. See Appendix B for aerial images of the aspects shown here.

3.3.4 Model Output – Vulnerability to Lugworm Collection

The vulnerability of sediment areas to lugworm harvesting were estimated for the whole of the BNNC EMS, acting as a predictor for areas where impacts are most likely to occur. The more vulnerable an area, the more likely negative impacts from lugworm collection will occur. The most vulnerable zones are areas which have been previously identified as suitable for collection, and additionally sensitive to harvesting disturbance. The model output map (Figure 3:9 a) depicts the sediment shores as varying levels of vulnerability, ranging from very low vulnerability to extremely high vulnerability.

Just over 5% of the sediment area was classified as having extremely high vulnerability, the locations of which can be seen in Figure 3:9 (b, c, d, and e). These zones included parts of Fenham Flats, Budle Bay, Newton Haven, and Boulmer. The most vulnerable sediment patches within Fenham Flats and Budle Bay ranged from very high to moderate suitability, and extremely high to high sensitivity. The most vulnerable sediment patches within Newton and Boulmer shores were classified as having extremely high suitability and low sensitivity. Within Fenham Flats and Budle Bay it is likely that individual collection events could cause greater and longer lasting impacts, but they are likely to occur less often. At Newton and Boulmer the impacts from each collection event may be smaller with faster recovery, but they are likely to occur more often, leading to a larger cumulative impact (Brown and Wilson, 1997).

Other areas with high or very high vulnerability classifications include: north and upper shore Boulmer, patches of Longhoughton, Newton lower shore, stretches of Seahouses, more of Budle Bay and Fenham Flats, Holy Island north shores, Berwick shores north of the pier, and the east side of the Eyemouth shore. Many of these identified vulnerable areas are currently protected by extant management e.g. the Lindisfarne NNR byelaws and other byelaws.
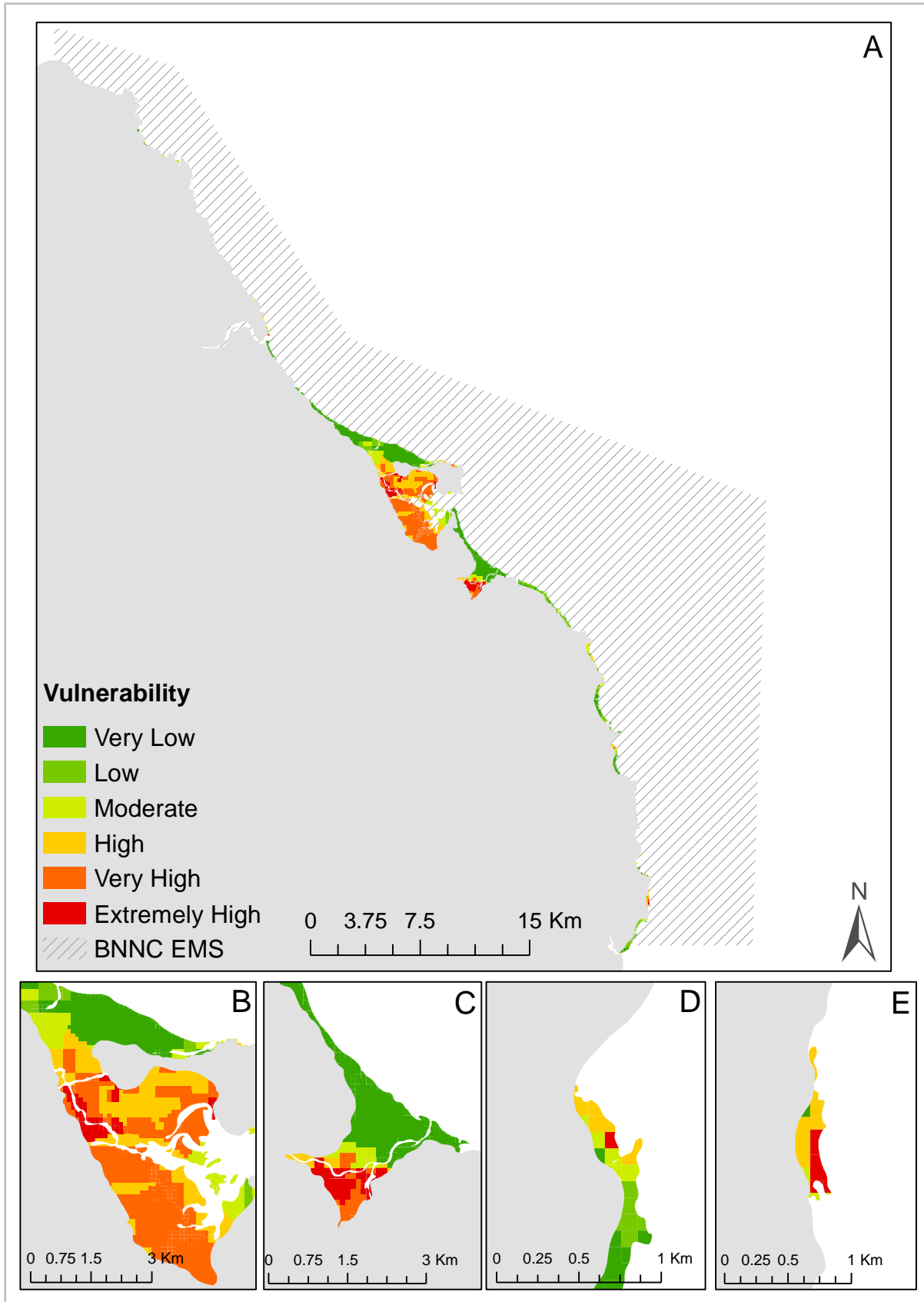
Figure 3:9: Vulnerability model output for the BNNC EMS (a), with the most vulnerable areas shown in greater detail: (b) Fenham Flats, (c) Budle Bay, (d) Newton, and (e) Boulmer. See Appendix B for aerial images of the aspects shown here.

3.4 Discussion

3.4.1 Lugworm Population Data

Field collected data for lugworm density, size, and species distribution, are the first broad scale data available on lugworm populations for the BNNC EMS, and appear to be the only lugworm population maps at such a large scale anywhere. The resulting maps for each lugworm criteria (especially species distribution and density) provide an extremely useful snapshot of the current lugworm populations within the BNNC EMS, with potential uses as part of future biodiversity assessments, resource management, biological reserve design, habitat management, species and habitat conservation planning, environmental risk assessments, population viability analysis, and community and ecosystem modelling (Franklin, 2010). Most importantly with regard to the aim of this thesis, the lugworm maps produced in this chapter have enormous potential to be used as a baseline for which to assess future change against, forming the basis for effective lugworm population monitoring, which may inform stock management in the future. The lack of historic lugworm population data locally was a major challenge in assessing the impacts of bait digging on the target species within the BNNC EMS (see Chapter 2). The supply of a broad scale baseline allows for the evidencing of change over time if overexploitation of stocks occurs in the future.

Lugworm distribution was patchy, with large variations both between and within shores, a common trait due to specific and complex habitat selection (e.g. Longbottom, 1970a; Flach and Beukema, 1994). The highest average density recorded (156 per m^2) was very high. Over 150 per m^2 has been recorded in previous studies (Nielsen et al., 2003), but is considered an extreme when compared to the typical range of between 3 and 80 worms per m^2 (Volkenborn and Reise, 2006, Cadée, 1976, Jones and Jago, 1993). In the Wadden Sea, where lugworm biomass is considered high, density is usually less than 50 per m^2 (Dankers and Beukema, 1983). Only 2.5% of the BNNC EMS data points contained an average lugworm density of over 50m^2, suggesting there are areas extremely well populated, but covering a relatively small area of the coast. The BNNC EMS appears to hold a considerable lugworm population.

Due to the use of the faecal cast size proxy to represent lugworm size, the data cannot be used to compare against directly measured lugworm sizes in other studies. However, it remains a useful tool for comparing sizes over time and position within the BNNC EMS. Black lugworm distribution was very patchy, with only a few shores recorded to hold this species (Eyemouth, Berwick, Boulmer, Newton, and North Sunderland), and logged at only 20 data points (0.5%). This suggests that for the vast majority of sediment area, lugworm collection can and will only occur via the traditional digging method, and not the less damaging bait pumping method (Fowler, 1999). It can be assumed that most of the harvesting falling within the BNNC EMS boundaries will be carried out by digging, and as such management planning should reflect this.

3.4.2 Model Outputs – Suitability, Sensitivity, and Vulnerability to Lugworm Collection

The most suitable shores for lugworm collection identified by the model agreed well with those previously identified as highly collected in shore observations (Chapter 2), suggesting that the measures of suitability can translate into actual shore use, and confirms chosen criteria and weightings were appropriate (Jorgensen, 2011). The more suitable an area is for collection within the model output, the higher the collection intensity is likely to be in reality. The most suitable score possible from the model design would be an area of sediment which has: high lugworm density, large lugworm size, black lugworm present, muddy sand or sandy mud, parking and population centres in close proximity, and no bait digging regulations in place. The most suitable zones identified for lugworm collection were areas of sediment shore at Berwick, Newton, and Boulmer. Of these, two shores (Boulmer and Newton) already have some level of bait digging legislation in place (UK Marine SACs Project, 2001a; NCAONB, 2009). Both remain popular collection shores despite regulation due to zoning allowing collection in some areas (Boulmer) or non-compliance (Newton – see Chapter 2).

The sensitivity model identifies the zones that would be most sensitive (larger and longer lasting impacts) to the disturbance created by lugworm collection (i.e. bird disturbance, eelgrass uprooting, infauna mortality, etc. (e.g. Evans and Clark, 1993; Ferns *et al.*, 2000; Mieszkowska, 2010)). The most sensitive score possible would be

an area of sediment which has: SPA designation, eelgrass present, mud, low lugworm density, small lugworm size, and a high degree of shore isolation. The most sensitive zones were areas of Budle Bay and Fenham Flats, where bait digging is banned outside of a small, less sensitive, section of Fenham Flats – known as the 'Voluntary Bait Digging Zone' (UK Marine SACs Project, 2001a; NCAONB, 2009). It appears that the most sensitive areas of the coast are protected from lugworm collection disturbance impacts under existing management plans as long as enforcement is adequate. Other slightly less sensitive areas (moderate sensitivity classification) are not similarly protected from bait digging, such as areas within: Fenham Flats digging zone, Seahouses, Beadnell, Howick, and southern Boulmer. Although identified as less sensitive, these areas can still suffer from bait digging impacts. In the future, management may be required to expand into these areas to protect sediment shores over a larger and more diverse geographic area.

The final model identifies the zones which would be most vulnerable to lugworm collection impacts, areas which are both suitable and sensitive to some degree (Roberts *et al.*, 2010). The most vulnerable zones identified included parts of Boulmer, Newton, Budle Bay, and Fenham Flats. The only extremely vulnerable area without bait digging legislation currently in place is at Boulmer, where the most vulnerable patch of sediment falls outside of the no-digging zone. Management plans may wish to consider extending the no-digging zone at Boulmer to cover the entire shore, protecting the most vulnerable areas of the coastline fully. The additional most vulnerable areas of Newton, Budle Bay and Fenham Flats are all no-digging areas in existing management plans including various byelaws (UK Marine SACs Project, 2001a; NCAONB, 2009). However, enforcement and compliance remain issues in some areas (Chapter 2 & NCAONB (2009)).

3.5 Conclusions

The models produced within this study supply local fisheries data to managers, with the aim of informing future management plans, and helping to evaluate current management measures. The modelling methods used were cost effective (Martin *et al.*, 2005), primarily utilising valuable existing data, literature review, and local expert knowledge, with a small amount of supplementary field data collection. There is scope for the models to be utilised and developed further for a variety of local intertidal fisheries in the BNNC EMS and beyond, supplying affordable data to marine managers. Model derived information, such as the outputs of this chapter, unquestionably contain a level of uncertainty (Cressie *et al.*, 2009), based on multiple assumptions, such as: existing data accuracy (e.g. habitat maps), expert opinion representativeness (e.g. suitability weightings), field data interpolation, and generalisation of the literature, etc. Resemblance of the suitability model outputs to the shore observation results from Chapter 2 alleviate some of the uncertainty and doubt, however, assumptions must be acknowledged by managers when analysing the model outputs.

Overall, this chapter has further revealed the spatial patterns of lugworm collection within the BNNC EMS, building on the shore observations in Chapter 2, and proving that simple and cost-effective modelling techniques can be extremely useful to managers. Designing the models has unravelled the motivations behind fishers selecting a target shore for lugworm harvesting, increasing the understanding of the fishery as a whole. Spatial modelling has proved an effective method to study intertidal collection, especially for unreported and relatively secretive fisheries where it can be difficult to obtain spatial data from more traditional methods such as interviews (e.g. McKay *et al.*, 1997), or vessel monitoring systems (e.g. Pedersen *et al.*, 2009). The models within this study have spatially defined the most suitable, sensitive, and vulnerable zones to lugworm collection within the study area, having the potential to direct management. It appears that existing management of lugworm collection spatially encompasses a good proportion of the most suitable, sensitive, and vulnerable areas identified by the spatial models. Berwick and south Boulmer are the major exceptions, where extremely and highly vulnerable areas are not currently protected from harvesting.

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Chapter 4 : Investigation of the Impacts of Lugworm Collection within the Berwickshire and North Northumberland Coast European Marine Site

4.1 Introduction

Impacts of fishing on marine ecosystems are well recognised and documented for fishing activities globally (e.g. Dayton *et al.*, 1995; Auster *et al.*, 1996; Thrush *et al.*, 1998; Turner *et al.*, 1999; Collie *et al.*, 2000; Coleman and Williams, 2002; Kaiser *et al.*, 2006b; Williams *et al.*, 2008; Smith *et al.*, 2011). However, intertidal fishing activities have received considerably less attention to date. The impacts of all fishing activities need to be understood if the global drive for biodiversity conservation is to be realised (Boonzaier and Pauly, 2016).

UK fisheries management requires the use of an evidence based method (Marine and Coastal Access Act, 2009); the approach to the management of commercial fisheries within European Marine Sites (EMS) was revised accordingly by DEFRA (MMO, 2014b, now referred to as the 'fishing in MPAs project'). The potential impacts of fishing activities are considered by conducting Habitats Regulations Assessments for each fishery-interest feature interaction within protected sites (MMO, 2014b). Fishing activities which are deemed to unfavourably affect site integrity are disallowed without adequate management measures. The impacts of intertidal collection activities on sand and mud flats were considered largely unknown in preliminary assessments, being identified as an area where additional empirical evidence is needed (MMO, 2014b). Management actions have already been taken for some fishing activities known to adversely impact interest features. Northumberland IFCA has for example, introduced two new byelaws within the BNNC SAC (NIFCA, 2016), to minimise impacts of mobile fishing gear on rocky reefs (e.g. Kaiser and Spencer, 1996; Kaiser *et al.*, 1998; Kaiser *et al.*, 2000; Hughes *et al.*, 2014), and bait digging on seagrass beds (e.g. Cabaço *et al.*, 2005; Mieszkowska, 2010; McCloskey and Unsworth, 2015; Silberberger *et al.*, 2016). Further management measures are possible as and when new evidence becomes available for fishery-interest feature interactions, an evidence base which this project hopes to contribute to. There is a need for site specific studies, relative to the local intensity and frequency of a fishing activity, to adequately inform managers whether fishing activities are compatible with the conservation objectives or designated features of MPAs, such as the BNNC EMS (Clarke and Tully, 2014).

Interest in intertidal fisheries impacts has increased in recent years, resulting in a growing body of literature (e.g. Ferns *et al.*, 2000; Kaiser *et al.*, 2001; Sheehan *et al.*,

2010; Erlandson *et al.*, 2011; Bertocci *et al.*, 2014; Clarke and Tully, 2014; Manríquez *et al.*, 2016; Toupoint *et al.*, 2016). Bivalve harvesting within soft sediment intertidal environments has received much attention (e.g. Ferns *et al.*, 2000; Dias *et al.*, 2008; Constantino *et al.*, 2009; Van Alstyne *et al.*, 2011; Ortega *et al.*, 2012; Lewis *et al.*, 2013; Boldina and Beninger, 2014; García-García *et al.*, 2015; O'Connell-Milne *et al.*, 2015), and our knowledge of bait digging for marine worms is not far behind (e.g. Blake, 1979b; Jackson and James, 1979; Shepherd and Boates, 1999; Skilleter *et al.*, 2006; Watson *et al.*, 2007; Mieszkowska, 2010; Pires *et al.*, 2012; Carvalho *et al.*, 2013; Mosbahi *et al.*, 2015; Watson *et al.*, 2017a; Watson *et al.*, 2017b). Within bait digging studies, lugworms are commonly studied in European contexts (e.g. Blake, 1979a; Shahid, 1982; Howell, 1985; Cryer *et al.*, 1987; Van den Heiligenberg, 1987; Beukema, 1995; Volkenborn and Reise, 2006; Volkenborn and Reise, 2007), with recent focus on large-scale or mechanical harvesting in vast areas such as the Wadden Sea (e.g. Van den Heiligenberg, 1987; Beukema, 1995; Volkenborn and Reise, 2007), In the UK, lugworm collection is primarily small scale, the effects of which have been investigated on the target species populations (Blake, 1979a; Shahid, 1982; Howell, 1985; Olive, 1993). Recent evaluations of small scale lugworm collection across the UK, specifically evidence of the effects on sediment communities as a whole, is lacking.

Both lugworm size and abundance can be altered by harvesting. Lugworm population structures can be altered by collectors preferentially removing the largest individuals (Shahid, 1982), and abundance can decrease substantially, from both removal and increased mortality of uncollected individuals (Beukema, 1995; Volkenborn and Reise, 2007). Where impacts are observed, recovery rates are variable between studies, ranging from one month to several years (Blake, 1979a; Cryer *et al.*, 1987; Beukema, 1995).

The physical disturbance of the sediment created by bait diggers can kill or damage infaunal species directly, or indirectly by creating conditions in which the organisms can no longer survive (Chandrasekara and Frid, 1998). Total infaunal biomass is often reduced after digging, with altered community structures due to the varying sensitivities of different species (Jackson and James, 1979; Van den Heiligenberg, 1987; Brown and Wilson, 1997; Watson *et al.*, 2017b). Digging disrupts sediment layering and alters the chemical concentrations in the sediment surface layer (Howell,

1985; Fowler, 1999), which can impact the organisms living within. The reduction of the target species after harvesting can also impact the infauna, especially when they are important in structuring the community (Cryer *et al.*, 1987; Lawton, 1994; Wright and Jones, 2006; Volkenborn *et al.*, 2007a; Volkenborn and Reise, 2007). There is evidence that removing lugworms and their bioengineering effects alters the sediment community structure, with different species reacting either positively or negatively to the altered habitat (Volkenborn and Reise, 2006; Volkenborn and Reise, 2007; Petrowski *et al.*, 2016; Whitton *et al.*, 2016; Sousa *et al.*, 2017). Recovery rates of infaunal communities after bait digging range from several months, up to 5 years for the most vulnerable species (Van den Heiligenberg, 1987; Beukema, 1995; Fowler, 1999).

The severity of impacts associated with bait worm collection is linked to the method and intensity of harvesting. Mechanical harvesting, which mainly occurs in the Wadden Sea, is the most disruptive method, with the most severe impacts observed (Van den Heiligenberg, 1987; Beukema, 1995). Bait dragging is another very disruptive method, primarily used for the collection of ragworms in Poole Harbour (Dyrynda, 1995; Underhill-Day, 2008; Birchenough, 2013). There is evidence that the intensity of hand collection, the most common collection method, is an important factor in determining the level of impacts upon the target species, with implications for management measures: low intensity collection resulted in no observable changes in abundance of *A. marina* (Blake, 1979a), whilst elsewhere on the same Northumberland coastline, overexploitation lead to a population crash (Olive, 1993). It is therefore important that the method and intensity of collection within studies are representative of the actual collection activities occurring in the areas where evidence is required. Impact strength is also site specific (Watson *et al.*, 2017b), leading to the requirement of localised assessments to accurately inform management.

There are two main methods used in fishing impact studies in the scientific literature: comparative and experimental (FAO, 2005). Both methods have their own advantages and limitations. Comparative studies compare sites of differing fishing intensities, with the state of the community indicating the impact of actual fishing events (FAO, 2005). However, it can be difficult to reliably quantify the fishing intensity at the scale of sampling, which can result in local heterogeneity or patchiness of fishing effort causing bias in results (Hughes *et al.*, 2014). Experimental studies measure the characteristics

of a site before and after controlled fishing events (FAO, 2005). This method is useful to observe the direct impacts from a known fishing intensity, however, the experimental study areas are usually unrepresentative of the scale of the fisheries – both spatially and temporally (Hughes *et al.*, 2014). Experimental studies of bait worm collection impacts use either simulated digging (e.g., Brown and Wilson, 1997; Griffiths *et al.*, 2006; Watson *et al.*, 2007; Carvalho *et al.*, 2013), or lugworm exclusion methodologies (e.g., Volkenborn and Reise, 2006; Volkenborn *et al.*, 2007a; Volkenborn and Reise, 2007; O'Brien *et al.*, 2009; Kuhnert *et al.*, 2010a; Lei *et al.*, 2010; Petrowski *et al.*, 2016). Simulated digging emulates the initial disturbance, whilst exclusion of lugworms explores the secondary impacts of the reduction in lugworms and their ecosystem engineering effects.

The aim of this chapter is to explore the impacts of lugworm harvesting within the BNNC EMS on the population size and structure of the target species, *Arenicola marina* and *Arenicola defodiens*, and the associated sediment community effects. Both comparative and experimental methodologies are used, to study both the direct impacts from known harvesting intensities, and observable impacts from actual fishing pressures, and how these relate to each other within the EMS. Density and mean size of the target species are determined at three shores of varying harvesting intensities, along with the overall sediment community structure, taxonomic richness, and abundances of individual infaunal species. Simulated digging and lugworm exclusion experiments are conducted within a single recently undisturbed site (within a protected area), with before, after, and subsequent recovery conditions explored for both the target species and the associated sediment community.

4.2 Methods

4.2.1 Comparative study

Site Selection

Three shores were required for comparison, each with a different level of collection pressure: no collection, low collection, and high collection (Figure 4:1). Shores within the BNNC EMS with appropriate collection pressures were identified on the basis of preliminary shore visits combined with advice from expert authorities (Angling Trust, Natural England, and the Northumberland Inshore Fisheries Conservation Authority) to establish known bait-digging activity. The selected shores were observed regularly from December 2013 to July 2014. Each site was visited at low tide 1-2 times per month throughout the monitoring period to estimate the intensity of lugworm collection occurring at each, validating the assumed collection pressure classifications. The observations were made on a mix of both weekdays and weekends, and under various environmental conditions (e.g. weather and seasons), to remove confounding effects presumed to influence bait digging behaviour (Fowler, 1999). At each visit, the number of collectors present at each site was recorded.

A section of Fenham Flats, Holy Island (O.S. Grid Reference NU121424), outside of the bait digging zone, was selected as the 'no collection' site (Figure 4:2), being a protected and actively enforced area. Newton Haven (O.S. Grid Reference NU243243) was chosen as the 'low collection' shore, due to anecdotal collection despite protection, and occasional enforcement. The southern half of Boulmer (O.S. Grid Reference NU267136) was selected as the 'high collection' shore, with intensive bait collection occurring, and no protection. All sites are rural with only small settlements or no settlements close by, and no obvious pollution sources. The main difference between sites is the slope, with Boulmer and Newton having a shallow sloping aspect towards the low water mark, compared to Fenham Flats which is more level.

The locations of each site in relation to the position within the BNNC EMS are shown in Figure 4:1, with aerial images of each site shown in Figure 4:3, depicting the habitat types and sampling areas more clearly.
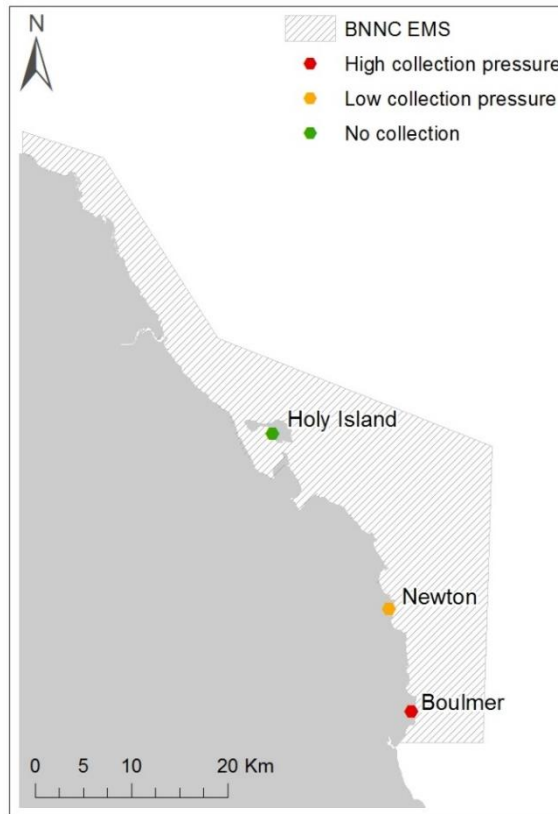
Figure 4:1: Locations of sample sites: Boulmer (high collection pressure), Newton (low collection pressure), and Holy Island (no collection), within the BNNC EMS.

Figure 4:2: The location of the bait digging zone at Holy Island (where bait digging is allowed), in relation to the sampling site selected as 'no collection'

Figure 4:3: Aerial images of each study site (Map data @2018 Google). A = Boulmer, B = Newton, C = Holy Island. Approximate sampling areas are shaded in grey.

Sampling

Sampling was carried out in March 2014, at low spring tides. At each shore, ten quadrats (1m^2) were placed randomly (random number sampling, with numbers generated equalling steps along the shore until the next sample) along the lower shore where bait digging primarily occurs (Fowler, 1999). Within each quadrat, *Arenicola* casts were counted and randomly selected subsamples of five casts per quadrat were measured for cast diameter, to the closest millimetre. *A. marina* and *A. defodiens* casts were grouped to give a single count or size measurement of 'lugworm casts'. Number

of casts can be used as a proxy for abundance (Flach and Beukema, 1994), and diameter of the individual cast strands can be used as a proxy for worm size (Retraubun *et al.*, 1996b). The use of these proxies allowed for effective and efficient sampling of lugworm populations whilst minimising sediment disturbance. Counting casts rather than individual worms is acknowledged to have an undercount issue, which was found to be 6% by Farke *et al.* (1979). However, no correction was performed in this study, as the aim is not to compare the lugworm populations to elsewhere in the world where actual counts have been conducted, but to compare different sites within this study, and to act as baseline data for future measurements locally, which should also use the cast count method to minimise disturbance.

Additionally, ten sediment cores (approximately 4,500 cm^3) were collected, using a post hole auger. This was screwed into the sediment to the required depth (30cm) before being extracted, retaining the sediment on the device. This method was efficient, especially in more muddy, or waterlogged areas, where box corers were unsuitable. The nature of *Arenicola* burrows, and the depth of bait digging trenches, required 30cm deep cores to permit observation of effects beyond the most populated surface sediment, including changes in species which live at depth, or preferentially in the lugworm burrows (e.g. flatworms (Reise, 1987; Reise, 2002)). Although standard intertidal sediment sampling procedure, smaller sample volumes/sizes are not best suited for collecting larger macrofauna (Eleftheriou and McIntyre, 2008). It is possible that the corer diameter of 15cm used in this study may underestimate the abundance of larger species, such as large bivalves, etc. which also happen to be some of the most vulnerable species to damage from bait digging (Jackson and James, 1979; Beukema, 1995). This limitation is acknowledged, but larger sample areas were not suitable for use within this study, especially within small experimental plots.

Sediment samples were immediately sieved onsite through 0.5mm mesh sieve bags. Material retained were transferred into screw top plastic bottles (800ml) with enough 70% ethanol to cover the samples for preservation. A further two sediment cores were collected at each site for Particle Size Analysis (PSA), which was carried out off site. PSA samples were dried overnight in a low temperature oven (approx. 100°C) (Poppe *et al.*, 2000), the particles gently separated, and 100g per sample passed through a series of sieves of decreasing mesh sizes using a sieve-shaker. The sieve sizes used

in micrometres were: 63, 125, 250, 500, 1000, and 2000. The resulting material retained in each sieve was weighed and recorded.

Faunal samples were stained using Rose Bengal solution in 70% ethanol, to distinguish biota from the inorganic material and accelerate sorting (Tagliapietra and Sigovini, 2010). After 3 days staining, samples were added to trays containing clean water. Organisms were sorted by eye, using fine metal forceps and pipettes, and transferred to 70% ethanol for further storage. Fixing in formalin was deemed unnecessary. Organisms were identified to species level where possible using a compound microscope. Exceptions were taxa such as Nematodes and *Capitella* spp., where separation to species level could not be justified due to the additional time resources required.

4.2.2 Experimental study

Site Selection

The site for simulated digging and exclusion experiments was required to be undisturbed within medium to long term time frames. Fenham Flats, at Holy Island (outside of the bait digging zone) was selected (Figure 4:4), as this area is protected and actively enforced by the Lindisfarne NNR wardens and manager, therefore was assumed to be largely undisturbed in recent years. Within Fenham Flats, possible sampling sites were further screened for suitability of field sampling (proximity to a water body for sieving on site, adequate distance from the bait digging zone, proximity to the causeway for accessibility and safety, isolation from walkers etc. for minimal experimental disturbance (Figure 4:5). An aerial image of the area can be seen in Figure 4:6, showing the habitat types more clearly.
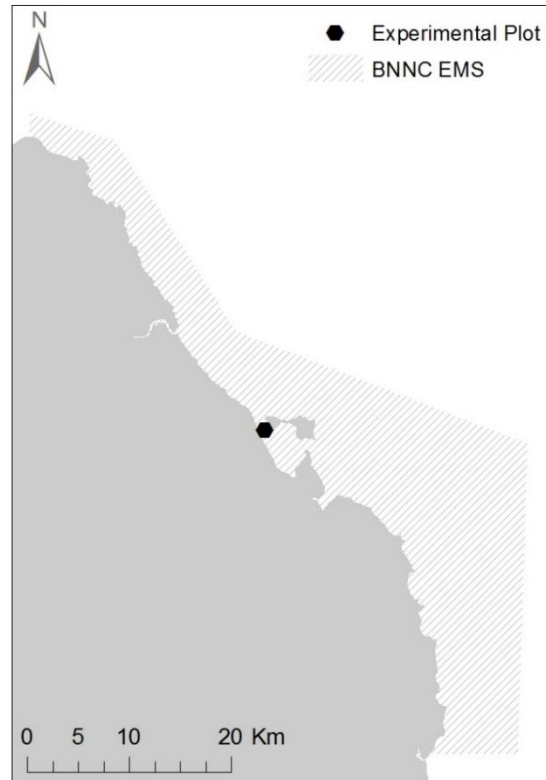
Figure 4:4: Location of the experimental site at Fenham Flats, within the BNNC EMS.
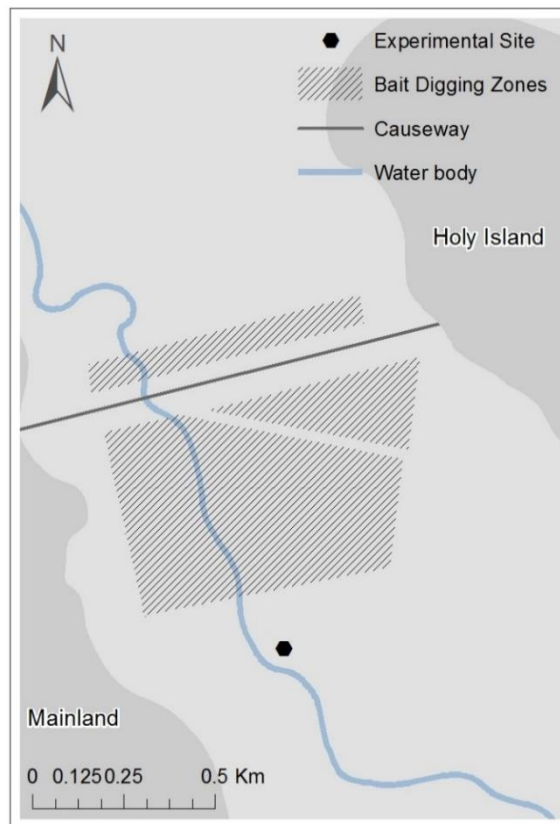
Figure 4:5: Experimental plot position at Fenham Flats, Holy Island, in relation to the causeway, water body, and bait digging zones.

Figure 4:6: Aerial image of the experimental study site (Map data @2018 Google). Approximate sampling area shaded in grey.

Experimental set-up

Sediment disturbance and associated reduction in lugworm abundance created by bait digging (e.g. Beukema, 1995; Fowler, 1999) was simulated within 25 4m^2 experimental plots, spaced 5m apart. These were marked out in two parallel lines with wooden posts marking each corner (Figure 4:7). Each plot was randomly assigned a treatment using a random number generator, and labelled accordingly. There were five different treatments, with five replicates of each.
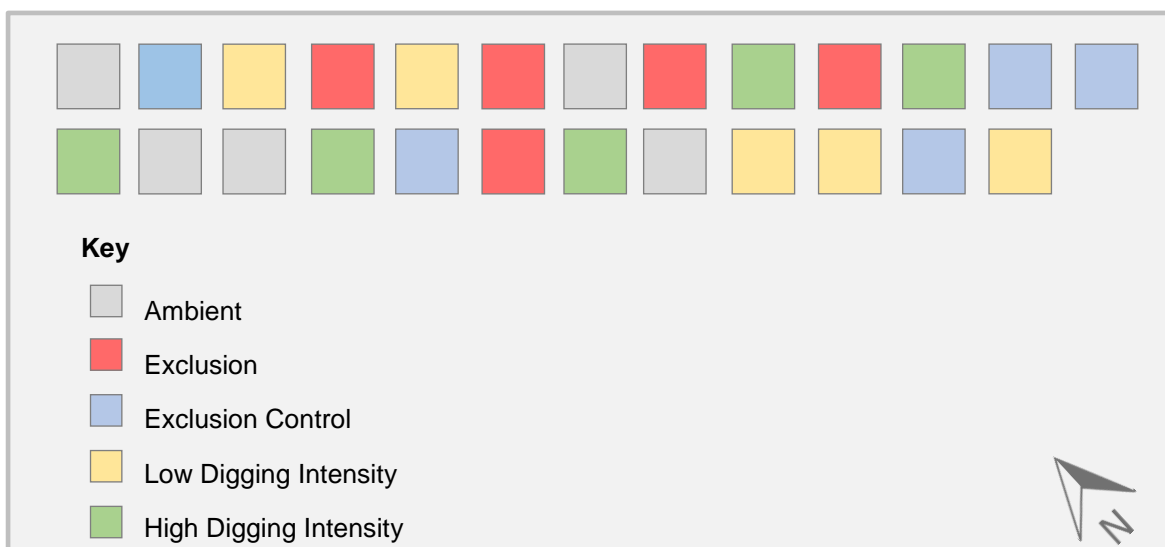
Figure 4:7: Experimental plot layout within the study site at Fenham Flats, Holy Island. Each plot is 4m^2, and spaced 5m apart (not drawn to scale – represents order of treatments only).

Ambient plots were left untouched as a control. Exclusion plots used 1mm mesh polyethylene nets, inserted horizontally, approximately 10cm deep into the sediment, to remove lugworms without disturbing the other fauna (a method previously used by: Volkenborn and Reise, 2006; Volkenborn *et al.*, 2007a; Volkenborn and Reise, 2007; O'Brien *et al.*, 2009; Kuhnert *et al.*, 2010a; Lei *et al.*, 2010). Exclusion control plots were similarly dug to 10cm, with no net inserted, controlling for the sediment disturbance caused when inserting an exclusion net. Low digging intensity plots were completely dug over to a minimum depth of 30cm once every three weeks, and plots backfilled, with no lugworms removed. The same digging methods were used for the high digging intensity plots, but with an increased frequency of once per week. All treatments ran for ten weeks (from 18th April 2014), with a subsequent recovery period left untouched for eleven weeks.

Sampling

All plots were sampled before treatments began, after 10 weeks of treatments, and again after recovery period of 11 weeks. Recovery sampling occurred for the control and simulated digging treatments only, the exclusion plots were not sampled again due to recovery requiring the removal of nets, which would have introduced a new disturbance.

Within each plot, *Arenicola* and the sediment communities were sampled using the same methodologies as described for the comparative study (for details see 2.2.1 Comparative Study – Sampling). *Arenicola* casts were recorded for the whole plot area (4m^2) rather than using quadrats, and three sediments cores were taken randomly within each plot. Sorting and identifying infaunal organisms within the sediment samples also followed the same methodology as previously described, as well as PSA of two further sediment samples (see 2.2.1 Comparative Study – Sampling).

Sediment conditions were recorded throughout the treatment and sampling regime. Changes in the surface sediment colouration were recorded at each site visit. Sediment penetrability was measured in each plot after the treatment period, by measuring the penetration depth (cm) of a garden fork dropped from 1m above the surface (adapted from Johnson *et al.* (2007)). Exclusion plots were not included in this analysis, due to the net affecting the depth the fork could penetrate.

4.2.3 Data Analysis

Univariate statistics were analysed using Minitab version 17, and multivariate with PRIMER software. Differences between sites, treatments, and times (before/after) were tested using ANOVA or Paired T-tests where parametric assumptions were met (normal distribution (or normalized using log or square-root transformations) and similar variances). Kruskal-Wallis or Mann-Whitney were used where normality assumptions could not be met (Underwood, 1997; Dytham, 2011). Subsequent pairwise comparisons were made where necessary for ANOVA tests (Tukey). PSA was graphically plotted, and each site classified into existing sediment type categories using granulometric types (EUNIS and Folk (1954)). Diversity was measured using the Shannon Wiener function (H), which was calculated for each sample and averaged for sites or treatments. Community structure was analysed using Bray Curtis Similarity (on square root transformed averaged data), with results expressed in Multidimensional scaling (MDS) plots. SIMPER analysis was used to determine the species responsible for the differences observed, which were subsequently plotted graphically.

4.3 Results

4.3.1 Collection Pressure and Sediment Characteristics

Bait collection observations at each comparative site (Boulmer, Newton, and Holy Island) validated the assumptions made from preliminary visits and expert advice. It was confirmed that Boulmer has a high collection pressure, Newton low collection, and Holy Island no collection (Table 4:1) occurring on observed dates.

Table 4:1: Validation of the collection pressure classifications assigned to each shore from observations recording the number of lugworm collectors present per shore visit (visited regularly between December 2013 and July 2014). Averages of collectors presented as means with standard deviation. Boulmer n=16, Newton n=7, Holy Island n=9.

Location	Collection Pressure	Average no. collectors per visit	S.D
Boulmer	High	6.56	9.76
Newton	Low	0.29	0.76
Holy Island	Not Collected	0	0

Sediment Particle Size Analysis (PSA) showed that overall, the sediment characteristics of the four sample sites (comparative and experimental studies) were largely similar to each other. Boulmer has the largest amount of fine particles (silt/clay), with 13% finer than 63 micrometres, and 54% finer than 125 micrometres, compared to less than 2% and 12% respectively at the other sites. Cumulative percentage sediment particle size data can be seen in Figure 4:8 for all sites. The PSA data were categorized further into three standard granulometric types (Folk, 1954): silt/clay (,63 micrometres), sand (63-2000 micrometres), and gravel/cobbles (>2000 micrometres), with the data shown in Table 4:2. All sites are predominantly sand, with all sites containing over 86% of this granulometric type (63-2000 micrometres). Despite this, Boulmer would narrowly classify as 'muddy sand' (being over 10% silt/clay), and all other sites as 'sand' in the common classification system designed by Folk (1954). These classifications would both be reclassified as 'sand and muddy sand' in the simplified EUNIS habitat classification system (Long, 2006).

Organic content of the three sites can be inferred from established relationships between organic content and particle size. There is a negative correlation between grain size and organic matter, due to the greater sorptive capacity of finer sediments (Dale, 1974; DeFlaun and Mayer, 1983; Mayer, 1993; Boudreau *et al.*, 2001).

Therefore it can be assumed that Boulmer has a higher organic matter content than Newton or Holy Island.
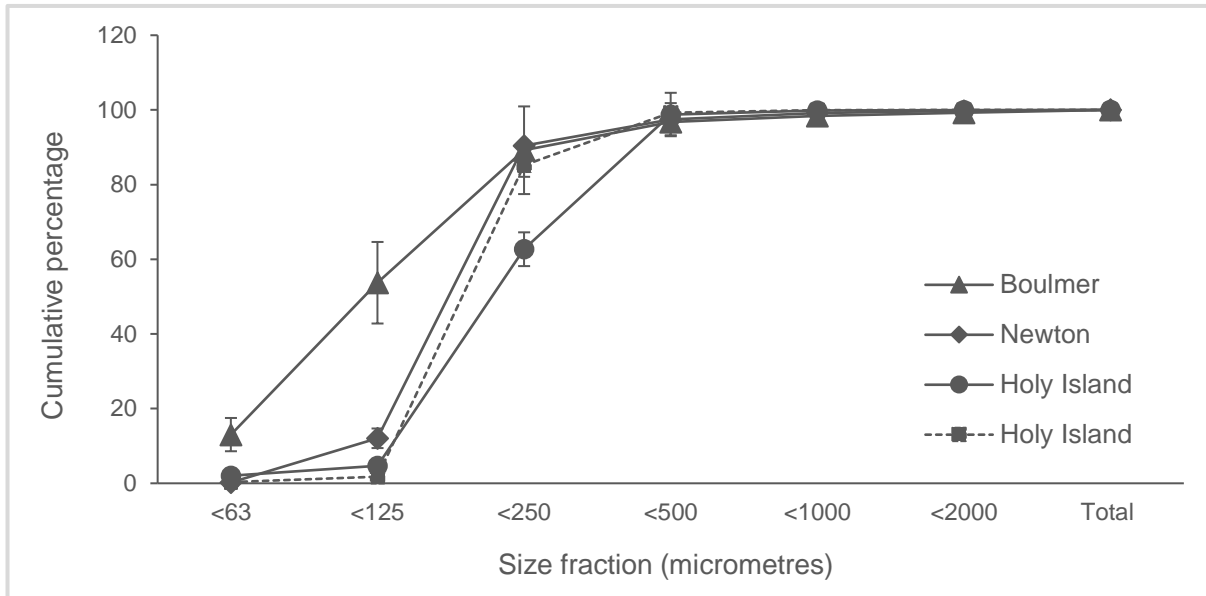
Figure 4:8: Average cumulative percentage (mean +/- SD) of the particle size in micrometres for all four study sites (comparative study = solid lines: Boulmer (high collection pressure), Newton (low collection pressure), and Holy Island (no collection); experimental study = dashed line: Holy Island). Samples were collected during March 2014 for the comparative sites, and April 2014 for the experimental site, using a core measuring 30cm deep and 15cm diameter (n=2 for all sites).

Table 4:2: Average percentage (mean ± SD) of the sediment samples made up of the three granulometric types (silt and clay = <63 micrometres; sand = 63-2000 micrometres; gravel and cobbles = >2000 micrometres), at each site (comparative study: Boulmer (high collection pressure), Newton (low collection pressure), and Holy Island (no collection); experimental study: Holy Island Experimental). Samples were collected during March 2014 for the comparative sites, and April 2014 for the experimental site, using a core measuring 30cm deep and 15cm diameter (n=2 for all sites).

Granulometric Type	Boulmer	Newton	Holy Island	Holy Island Experimental
Silt/Clay	13.04 (± 8.91)	0.23 (± 0.28)	2.05 (± 1.17)	0.32 (± 0.10)
Sand	86.18 (± 8.56)	99.57 (± 0.01)	97.92 (± 1.36)	99.68 (± 1.37)
Gravel/cobbles	0.77 (± 0.40)	0.20 (± 0.17)	0.03 (± 0.01)	0 (± 0)

4.3.2 Comparisons between sites with differing collection pressures

Target species

The mean densities of *Arenicola* spp. per quadrat ($1m^2$) are significantly different between sites (ANOVA, F = 9.78, df = 2, 27, P <0.001). It was revealed by *post hoc* Tukey pairwise comparison (P = 0.05) that lugworm density was significantly lower at the uncollected site, Holy Island (mean = 13.4 ± 5.27 SD), whilst Boulmer and Newton (collected sites) had statistically similar densities. The mean densities for all sites can be seen in Figure 4:9. Figure 4:10 shows the median lugworm cast diameters at each site, which do not statistically differ between shores (Kruskal-Wallis, H = 1.32, df = 2, *P* > 0.5).
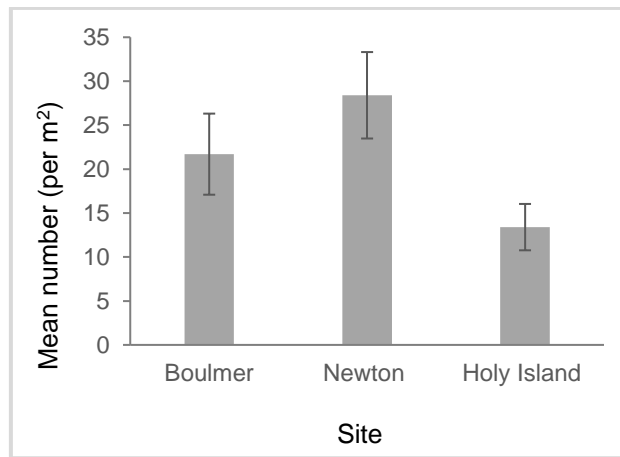
Figure 4:9: Mean (± SD) number of lugworms per m² from three sites of varying collection pressure (Boulmer = high collection pressure, Newton = low collection pressure, Holy Island = no collection), sampled March 2014, using quadrats (1m²) to count casts on the surface; n = 10 for all sites.
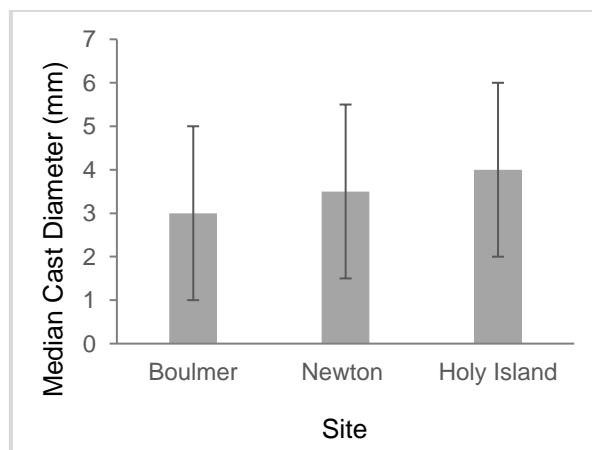
Figure 4:10: Median (± range) cast diameters (mm) of lugworms from three sites of varying collection pressure (Boulmer = high collection pressure, Newton = low collection pressure, Holy Island = no collection), sampled March 2014, with 5 casts measured from each 1m² quadrat (10 per site); n = 50 for all sites.

Infaunal Community

The total abundance of the infaunal species and taxa recorded at each site for all samples combined is seen in Table 4:3. Annelids dominate at all three sites, with crustaceans also occurring in high numbers. The three most abundant taxa recorded were: *Notomastus latericeus*, *Tubificoides* sp., and *Urothoe poseidonis*. Both *N. latericeus* and *Tubificoides* were only present in high numbers at Holy Island, whilst *U. poseidonis* were much more abundant at Newton.

Some key prey species for wading birds include *Cerastoderma edule*, *Limecola balthica, Peringia ulvae, Corophium volutator, Alitta virens,* and *Lanice conchilega,* along with smaller oligochaetes, polychaetes and molluscs (Smith and Evans, 1973; Goss-Custard *et al.*, 1977; Hicklin and Smith, 1984)*.* Most of these also happen to be some of the largest size taxa recorded within the study sites. The mean abundances of these important infaunal species at each site can be seen in Figure 4:11. Holy Island contains the highest average and total abundance of most of these species, apart from *Lanice conchilega* which was far more abundant at Boulmer (Figure 4:11 and Table 4:3).

Table 4:3: Total number of infaunal species and taxa within sediment samples (4,500 cm^3) collected from three shores of differing collection pressures (Boulmer = high collection pressure, Newton = low collection pressure, Holy Island = no collection). Samples were collected in March 2014 on low water spring tides (n=10 for all shores).

Species/Taxa	Boulmer	Newton	Holy Island
ANNELIDA			
Arenicola sp.	10	7	8
Capitella sp.	6	0	13
Enchytraeidae indet.	0	0	5
Eteone longa	2	2	7
Eumida sp.	0	3	0
Harmothoe sp.	1	0	0
Lanice conchilega	40	1	0
Magelona sp.	1	4	1
Malmgrenia sp.	2	0	1
Nephtys sp.	1	1	0
Alitta virens	0	0	3
Notomastus latericeus	1	0	141
Paraonis fulgens	0	43	70
Phyllodoce mucosa	29	3	1
Pygospio elegans	25	0	6
Scolelepis foliosa	0	1	1
Scolelepis squamata	0	1	1
Scoloplos armiger	36	8	33
Spio martinensis	7	74	3
Spiophanes bombyx	1	1	7
Tubificoides benedii	0	0	13
Tubificoides sp.	0	1	150
CRUSTACEA			
Allomelita pellucida	2	0	1
Ampelisca brevicornis	3	0	0
Bathyporeia sp.	0	2	1
Bathyporeia elegans	0	8	0
Urothoe poseidonis	30	194	8
Dexamine sp.	0	1	0
Corophium volutator	0	0	2
Idotea balthica	1	0	0
Ostracoda indet.	0	1	0
Pontocrates arenarius	0	1	0
Monopseudocuma gilsoni	0	5	0
Tanaissus lilljeborgi	0	17	0
Carcinus maenas	1	0	0
MOLLUSCA			
Cerastoderma edule	0	0	2
Limecola balthica	1	0	4
Scrobicularia plana	0	0	4
Fabulina fabula	6	9	0
Ensis siliqua	0	1	0
Peringia ulvae	0	0	7
NEMERTEA			
Unidentified	0	0	1
NEMATODA			
Unidentified	5	1	58
PREAPULA			
Priapulus caudatus	1	0	0
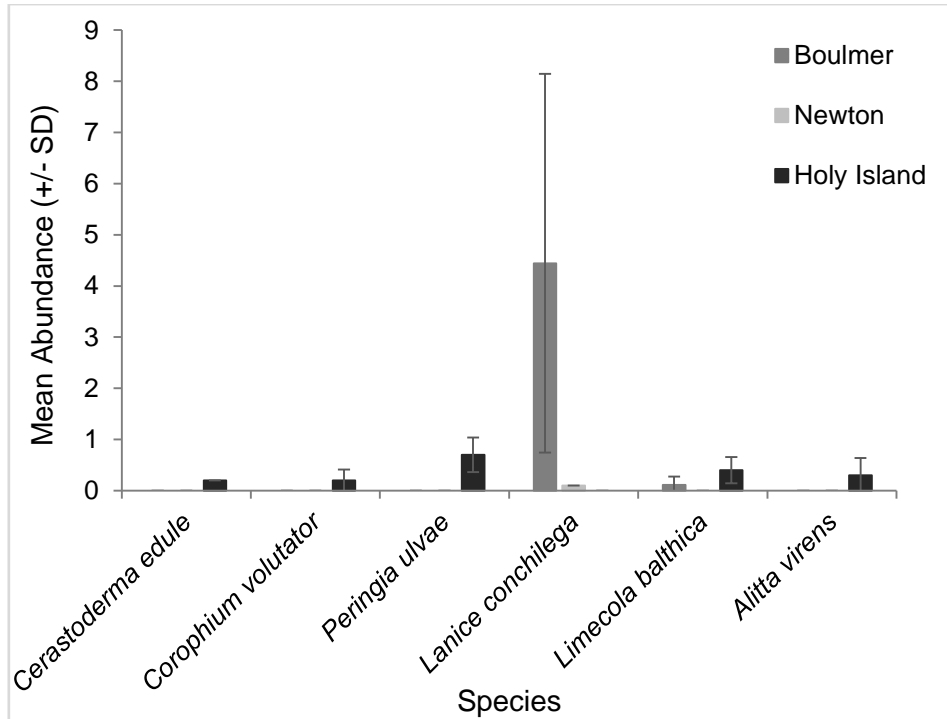
Figure 4:11: Mean (+/- SD) abundances of important wading bird prey species at each study site (Boulmer, Newton, and Holy Island). n = 10 for all sites.

The mean taxonomic richness is significantly different between shores (ANOVA, F = 3.53, df = 2, 28, P < 0.05). Holy Island, the uncollected site, had the highest mean taxonomic richness (mean= 11.4 ± 3.43 SD), whilst the lowest was Newton, the low collection pressure site (mean= 7.9 ± 2.60 SD) (Table 4:4). The median infaunal abundances were significantly different between sites (Kruskal-Wallis, H = 6.40, df = 4, P < 0.05) (Table 4:4), with decreasing abundances with increasing collection pressure. Boulmer, the high collection pressure site, had considerably lower average infaunal abundance (median = 20.0 ± 33.0 range), less than half the other sites. Despite the reduction in infaunal abundance with bait digging pressure, the diversity, as estimated by Shannon's diversity index, is not negatively impacted (Table 4:4).

Table 4:4: Median (± range) infaunal abundance, and mean (± SD) taxonomic richness and Shannon's diversity for each site with differing collection pressures (Boulmer = high collection pressure, Newton = low collection pressure, Holy Island = no collection), sampled March 2014 (n=10).

	Boulmer	*Newton*	*Holy Island*
Abundance	20.0 (± 33.0)	42.5 (± 54.0)	49.0 (± 77.0)
Taxonomic richness	9.0 (± 2.92)	7.9 (± 2.60)	11.4 (± 3.43)
Diversity	1.8 (± 0.39)	1.4 (± 0.23)	1.8 (± 0.33)

The community structure of the infaunal organisisms between sites is significantly different (ANOSIM: Global R=0.906, p=0.1%). Bray Curtis similarity shows that all shores have a comparable similarity level at around 40%. The Multi-Dimensional Scaling (MDS) plot of the Bray Curtis similarity (Figure 4:12) for the infaunal communities showed good discrimination between communities from each site, with the 25% similarity grouping overlay revealing higher similarity between the two collected sites (Boulmer and Newton) than the uncollected site (Holy Island). SIMPER analysis shows that the main species (greatest % contribution) responsible for the significant differences observed in community structure between the three sites are: *Urothoe poseidonis, Tubificoides sp., Spio martinensis,* and *Notomastus latericeus;* which are also some of the most dominant species recorded. The total abundances of each species from the SIMPER analysis is displayed in Figure 4:13. The uncollected site (Holy Island) contained the vast majority of *Notomastus latericeus and Tubificoides sp.* specimens*,* whilst *Urothoe poseidonis and Spio martinensis* were most abundant at the low collection pressure site (Newton).
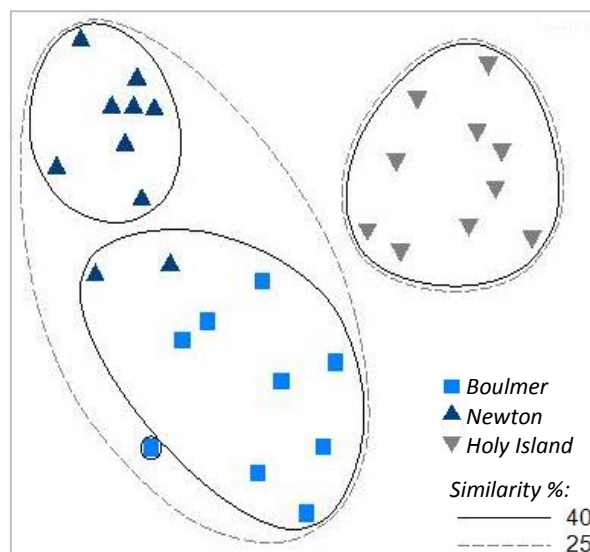
Figure 4:12: Non-metric multidimensional scaling (MDS) ordination of the Bray Curtis similarity based on square root-transformed averaged abundance data of the infaunal community from sites with differing collection pressures (Boulmer = high collection pressure, Newton = low collection pressure, Holy Island = no collection), sampled March 2014. 2D Stress: 0.12. Overlays of Bray Curtis similarity groupings at 25 and 40%.
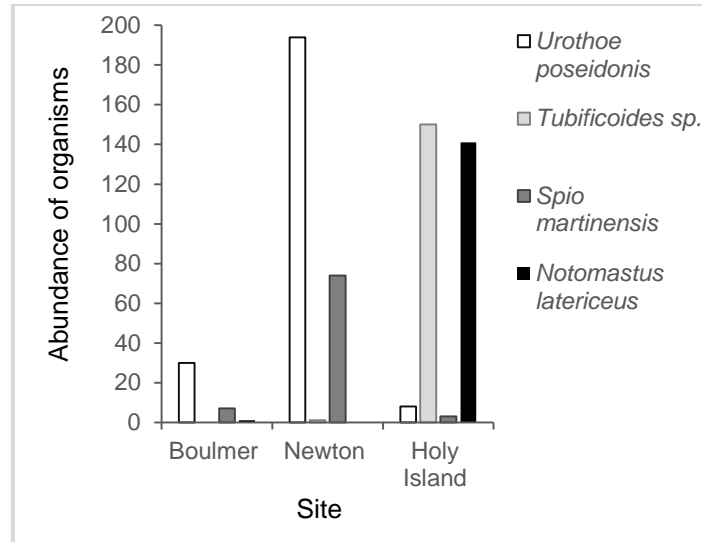
Figure 4:13: Total abundances of the four species most responsible for the difference in community structure per site (Boulmer = high collection pressure, Newton = low collection pressure, Holy Island = no collection), sampled March 2014 (n = 10 sediment samples each *4,500 cm³*).

4.3.3 Simulated Digging and Exclusion Experiments

The experimental study at Fenham Flats revealed significant effects of simulated bait collection activities, both between treatments, and over time (before, after, and recovery).

Target Species

Lugworm density was significantly different between treatments after ten weeks (ANOVA, $F = 64.24$, df = 4, 24, $P < 0.001$). *Post hoc* Tukey pairwise comparison ($P = 0.05$) showed that lugworm density was significantly lower for all treatments when compared to the ambient plots (Figure 4:14). Exclusion plots were designed to remove the majority of lugworms, however they only reduced the mean density to 65% of the ambient levels. High digging intensity plots had the lowest density, with just 13% of the ambient levels (despite no worms being removed in the treatment design).
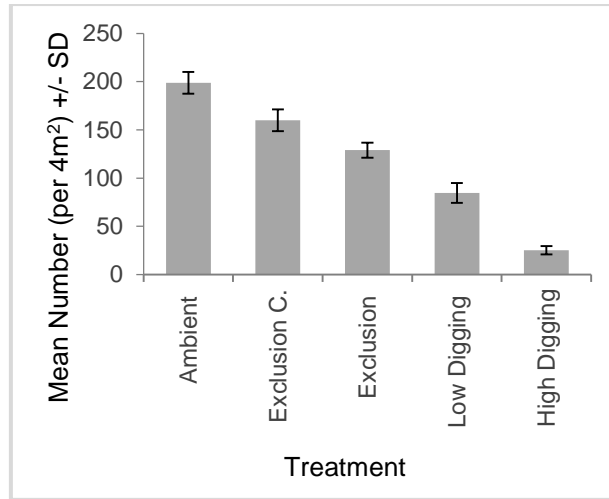
Figure 4:14: Mean (± SD) number of lugworms per plot (4m²) from each of five treatments (ambient, exclusion control, exclusion, low digging intensity, and high digging intensity), sampled after 10 weeks of treatment (June 2014) by surface cast counts; n = 5 for all treatments.
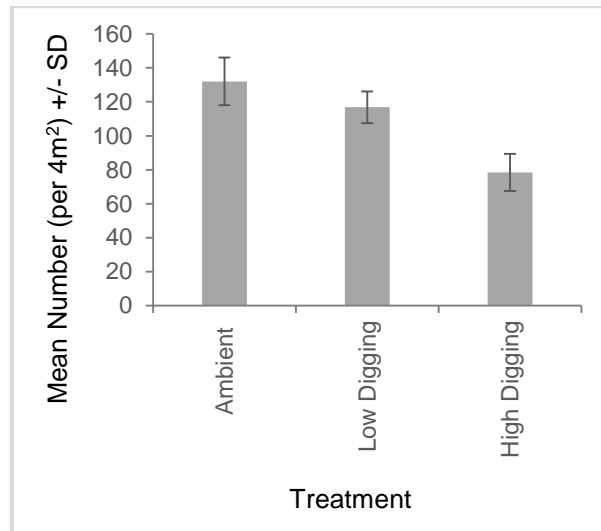
Figure 4:15: Mean (± SD) number of lugworms per plot (4m²) from each of three treatments (ambient, low digging intensity, and high digging intensity), sampled after 11 weeks of recovery (September 2014) by surface cast counts; n = 5 for all treatments.

After a recovery period of eleven weeks, lugworm density remained significantly different between treatments (ANOVA, F =7.04, df = 2, 12, P < 0.01). Tukey pairwise comparison (*P* = 0.05) showed that average lugworm density was only significantly lower in the high digging intensity plots (mean = 21.94 +/- 10.97 SD) when compared to the ambient conditions (mean = 28.12 +/- 14.06 SD) (Figure 4:15).

Infaunal Community

Table 4:5 displays the total abundance of the infaunal species and taxa recorded after ten weeks of treatments, within plots of each treatment, for all samples combined. The two most abundant taxa recorded were Nematoda, and *Pygospio elegans*, both of which decreased in abundance with the presence of digging. The response from *P. elegans* was more severe, decreasing from a total of 748 in the ambient treatments, to just 32 in the high digging intensity plots (Table 4:5), just 4% the unimpacted abundance.

Key wading bird prey species (*Cerastoderma edule*, *Limecola balthica, Peringia ulvae, Corophium volutator, Alitta virens,* and *Lanice conchilega*) also differ between treatments. The mean abundances of these important infaunal species at each site can be seen in Figure 4:16. These species had low total and mean abundances in all treatments, but were generally lowest in the digging treatment plots (Figure 4:16).

Table 4:5: Total abundance of infaunal species and taxa within sediment samples (4,500 cm³) collected from plots after 10 weeks of five different treatments (ambient, exclusion control, exclusion, low digging intensity, and high digging intensity). Samples were collected in June 2014 on low water spring tides (n=15 for all treatments).

Species/Taxa	Ambient	Exclusion Control	Exclusion	Low Digging	High Digging
ANNELIDA					
Arenicola sp.	16	19	16	16	6
Capitella sp.	31	35	16	28	9
Enchytraeidae indet.	30	34	43	49	5
Eteone longa	10	15	6	2	0
Alitta virens	17	19	19	11	4
Notomastus latericeus	1	0	0	0	0
Ophelia rathkei	15	31	25	20	4
Paraonis fulgens	6	6	1	0	1
Psammodrilus balanoglossoides	1	0	0	0	0
Pygospio elegans	748	570	333	80	32
Scolelepis squamata	2	1	0	1	0
Scoloplos armiger	8	2	4	2	1
Sphaeropsis sp.	0	1	0	0	0
Spiophanes bombyx	1	0	2	0	0
Tubificoides sp.	23	22	18	22	9
CRUSTACEA					
Bathyporeia pilosa	1	0	0	0	0
Bathyporeia sarsi	3	6	2	9	2
Urothoe poseidonis	31	13	6	1	0
Corophium volutator	4	3	24	0	0
MOLLUSCA					
Cerastoderma edule	3	1	3	1	0
Limecola balthica	7	4	5	2	3
Scrobicularia plana	10	8	16	2	6
Fabulina fabula	1	1	1	2	0
Mytilus edulis	1	0	1	1	0
Peringia ulvae	0	1	0	0	2
Retusa truncatula	1	1	0	3	1
NEMERTEA					
Unidentified	9	13	7	9	2
NEMATODA					
Unidentified	405	324	408	396	100
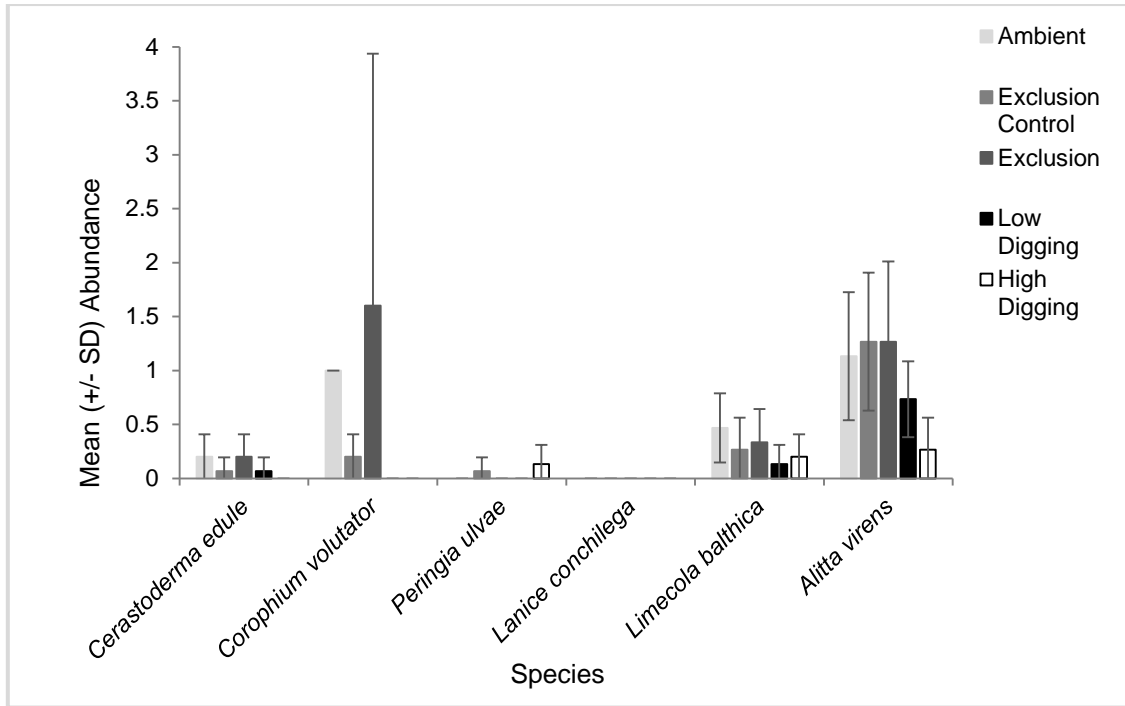
Figure 4:16: Mean (+/- SD) abundances of important wading bird prey species after 10 weeks for each experimental treatment (Ambient, Exclusion Control, Exclusion, Low Digging, and High Digging). n = 15 for all treatments.

The average taxonomic richness significantly differs between the treatments (Kruskal-Wallis, H = 38.49, df = 4, $P < 0.001$), with average taxonomic richness reduced in the exclusion, low digging intensity, and high digging intensity plots (Table 4:6). Mean infaunal abundance was also affected by treatment (ANOVA, F = 22.65, df = 4, 70, $P < 0.001$), with Tukey analysis revealing that only simulated digging treatments (low and high digging intensities) were significantly different from ambient, supporting significantly lower infaunal abundances (Table 4:6). Despite a reduction in taxonomic richness and infaunal abundance observed for the simulated digging treatments, diversity (Shannon's index) was not similarly effected by the presence of disturbance (Kruskal-Wallis, H =4.92, df = 4, 4 $P > 0.1$) (Table 4:6).

Table 4:6: Mean (± SD) infaunal abundance, and median (± range) taxonomic richness and Shannon's diversity for each treatment after 10 weeks of simulated disturbance (ambient, exclusion control, exclusion, low digging intensity, and high digging intensity), sampled June 2014 (n=15).

	Ambient	*Exclusion Control*	*Exclusion*	*Low Digging*	*High Digging*
Abundance	92.3 (±54.9)	75.5 (±44.7)	63.8 (±30.7)	43.8 (±23.4)	12.6 (±7.9)
Taxonomic richness	11 (±6)	12 (±8)	8 (±9)	9 (±7)	5 (±6)
Diversity	1.4 (±1.0)	1.5 (±1.0)	1.4 (±0.8)	1.4 (±1.2)	1.3 (±1.8)

Community structure also differed between treatments after ten week. MDS plot of the Bray Curtis similarity for the infaunal communities shows good visual discrimination between some experimental treatments, with the high digging intensity treatment well distinct from the others (Figure 4:17). ANOSIM (at 9999 permutations) reveals that community assemblages are statistically different between all treatments apart from exclusion control and ambient (R=0.364; p<0.01). The data for the high digging intensity treatment was analysed further to reveal which taxa were most responsible for the differences before and after. SIMPER analysis showed the taxa which contributed most to the differences were: *Pygospio elegans, Nematoda, and Tubificoides sp.* which were also dominant. The mean abundances of these taxa can be seen in Figure 4:18. The reductions for *Pygospio elegans* (paired t-test, t = 5.39, df = 14, P < 0.0001) and *Tubificoides sp.* (Mann-Whitney U-test, U = 288, $n_{1,2}$ = 15, P < 0.015) were significant, whilst Nematoda (paired t-test, t = -1.35, df = 14, P > 0.15) was statistically similar. SIMPER analysis was repeated to take into account the rarer taxa/species (using presence/absence data); the main contributing species were *Ophelia rathkei, Urothoe poseidonis, and Paraonis fulgens*, which were all markedly reduced. Out of a total of 28 taxa recorded in the high digging intensity plots, 23 were reduced after the disturbance period. The dominant taxonomic group by abundance was altered, from Annelids (71% before, 33% after), to Nematoda (17% before, 53% after).
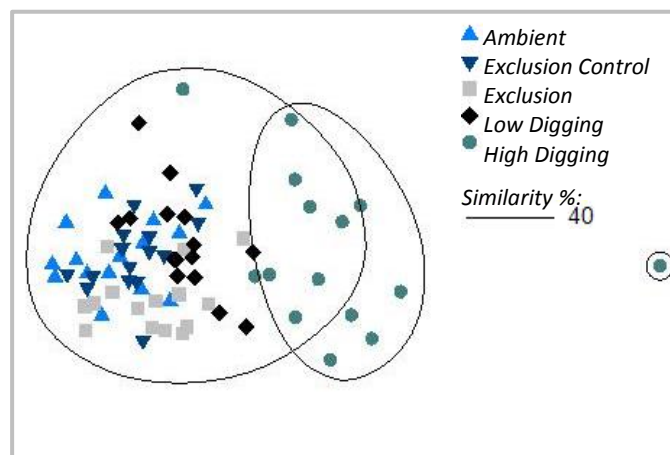
Figure 4:17: Non-metric multidimensional scaling (MDS) ordination of the Bray Curtis similarity based on square root-transformed averaged abundance data of the infaunal community from plots with differing treatments (ambient, exclusion control, exclusion, low digging intensity, and high digging intensity), sampled June 2014. 2D Stress: 0.19. Overlay of Bray Curtis similarity grouping of 40%.
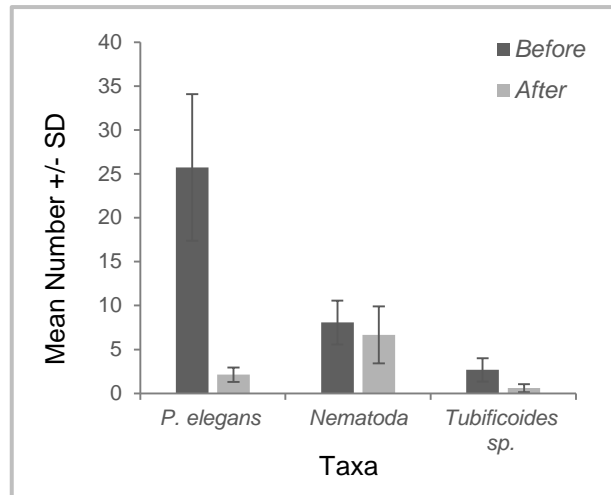
Figure 4:18: Mean (± SD) number (in 4,500 cm³ sediment samples) of the three taxa most responsible for the difference in community structure per treatment (ambient, exclusion control, exclusion, low digging intensity, and high digging intensity), before and after 10 weeks of treatment. Sampled June 2014 (n = 15).

After eleven weeks of no further disturbance, the infaunal community recovered well. Mean abundances of infaunal organisms for the ambient and simulated digging treatments can be seen in Figure 4:19 for all three sample periods: before, after, and after recovery period. Differences were no longer significant after the recovery period (ANOVA, F =0.28, df = 2, 42, P > 0.7). The mean taxonomic richness was also similar between treatments after the recovery period, with the high digging intensity plots having the highest average richness. Community structure also recovered; ANOSIM (at 999 permutations) reveals that community assemblages were no longer statistically different between treatments (R=0.098; p>0.1), and Bray Curtis similarity revealed a similarity level of >85%. MDS of the Bray Curtis similarity can be seen in Figure 4:20, with all treatments falling within the 40% similarity grouping overlay (apart from one outlier).
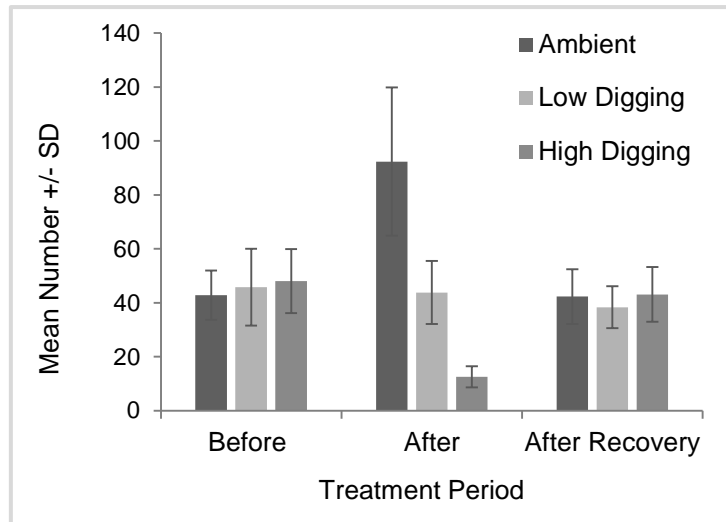
Figure 4:19: Mean (± SD) number of organisms (in 4,500 cm³ sediment samples) for 3 treatments (ambient, low digging intensity, and high digging intensity) before the simulated disturbance began, after 10 weeks of disturbance, and after a recovery period of 10 weeks (no further disturbance). n = 15 for all.
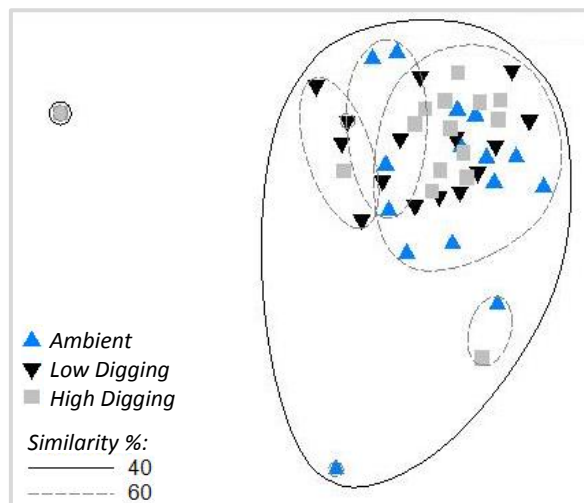
Figure 4:20: Non-metric multidimensional scaling (MDS) ordination of the Bray Curtis similarity based on square root-transformed averaged abundance data of the infaunal community from different treatments (ambient, low digging intensity, and high digging intensity) after a recovery period of 11 weeks. 2D stress = 0.22. Overlay of Bray Curtis similarity groupings at 40% and 60%.

Habitat Alterations

Sediment characteristics were noticeably altered during treatments. Simulated digging plots were darker from the redistribution of anoxic sediment to the surface. These alterations were still visible until the next disturbance (three weeks for low digging intensity). Sediment penetrability and softness was altered by digging, with fork penetration depths being significantly different between treatments (Kruskal-Wallis, H = 29.75, df = 2, P < 0.001), increasing with the presence of digging.

4.4 Discussion

Lugworm populations are maintained at current lugworm harvesting levels within the BNNC EMS. There is no evidence of reduced abundance or size at heavily collected sites, with populations sustained at harvestable levels despite long-term collection. Negative secondary impacts were observed, with the sediment communities altered by the presence of digging. Infaunal abundance was markedly reduced by the sediment disturbance associated with digging (in both comparative and experimental studies), the severity of such impacts linked to the intensity of collection. Recovery of experimental plots was rapid, suggesting that recovery on heavily exploited BNNC EMS shores may be possible if sufficient and well-timed no-take periods occurred. Managers must consider whether the level of impact observed is important at the EMS scale, which is discussed.

4.4.1 Impacts upon the target species – Lugworms

Neither lugworm density nor size appear to be correlated to long term bait digging pressures at current BNNC EMS exploitation levels, with high lugworm densities recorded at the intensively collected site, and no significant differences in size. In contrast, short term impacts were observed in disturbance experiments, with significantly reduced lugworm abundance recorded in the simulated digging plots.

The distinct scales, both spatially and temporally, of each study is the likely cause of dissimilarities, with the importance of representative scales highlighted in previous studies (Thrush *et al.*, 1996; Reise *et al.*, 2001; Watson *et al.*, 2017b). The small scale and short term experiments at Fenham Flats may have shown exaggerated impacts compared to the comparative study due to the nature of the disturbance and study site. Untouched sediment patches between and around experimental plots allowed for areas of 'undisturbed' sediment in close proximity. It is well known that lugworms have particular sediment requirements (Callame, 1961; Bruce *et al.*, 1963; Longbottom, 1970b), and it is possible that lugworms which did not suffer mortality from the digging disturbance (Hall, 1994; Beukema, 1995; Brown and Wilson, 1997) migrated out of disturbed plots with less suitable habitat (e.g. higher penetrability, restricted oxygen contact (Longbottom, 1970a), and reduced organic matter (Watson *et al.*, 2017b)) into the undisturbed areas. These substantial undisturbed areas would unlikely occur on fished sites, such as Boulmer, resulting in less lugworm migration

from collected areas, and maintained densities in dug zones. Additionally, the observable sediment alterations from digging (e.g. uneven surfaces, or discolouration of surface sediments (Watson *et al.*, 2017b)) were more severe and long lasting at the experimental site (personal observation: >3weeks vs 1 tide at Boulmer), due to lower wave energy slowing sediment recovery (Fowler, 1999; Reise, 2001; Watson *et al.*, 2017b), which could have further exaggerated the lugworm response to digging events.

Targeting digging within small experimental plots surrounded by large 'refuge' areas is not representative of the lugworm fishery within the BNNC EMS as a whole, but was important to consider and investigate to alleviate the interference of natural and other anthropogenic derived variability between sites in the comparative study. Lugworm densities can vary considerabley between locations (Cadée, 1976; Jones and Jago, 1993; Nielsen *et al.*, 2003; Volkenborn and Reise, 2006), with a lot of variation even within a geographically close area (Dankers and Beukema, 1983), often dependant on environmental factors such as food availability or sediment characteristics (Callame, 1961; Longbottom, 1970b; Groenendaal, 1979; Flach and Beukema, 1994; Kristensen, 2001; Reise *et al.*, 2001). Anthropogenic factors capable of influencing lugworm populations other than harvesting include trampling (Rossi *et al.*, 2007) and pollution (Matthiessen and Thain, 1989; Browne *et al.*, 2013). Small sediment differences between comparatvie sites (Boulmer being muddier and having higher orgainc content), and possible other unidentified anthropogenic impacts, could be capable of masking low level impacts of harvesting on lugworm populations. It is possible that lugworm populations have been negatively impacted within the BNNC EMS, but at levels which are not significant over natural and anthropogenic variability between sites.

Boulmer has maintained a high abundance and large average size of lugworm despite long term high intensity harvesting, suggesting little impact on the target species at current levels. This maintenance is likely due to *Arenicola* spp. ability to recolonise rapidly from both adult migration and larval recruitment (e.g. Blake, 1979a; Rees and Eleftheriou, 1989; Olive, 1993). Long-term population stability is enhanced by the long life-span of lugworms, and the inverse relationship between the rate of recruitment and adult density (Beukema and De Vlas, 1979; Farke *et al.*, 1979). Lugworm larvae's high dispersive potential (Günther, 1992; Tyler-Walters and Arnold, 2008) could be

masking local overexploitation, with recruitment from surrounding undisturbed areas helping to keep exploited populations stable in the long term. As long as larval supply is high, it appears that heavily exploited stocks can be sustained at harvestable levels over many decades.

Earlier studies have revealed reduced lugworm abundance due to harvesting (Beukema, 1995; Volkenborn and Reise, 2007), however these studies generally had a higher level of collection than that at Boulmer, with either simulated digging (more targeted disturbance), or mechanical harvesting (more disruptive). Shahid (1982) found no change in lugworm abundance with the presence of bait collection, but did record a reduction in size. Contrasting results in various studies reiterates the importance of resident studies to appropriately inform managers of the impacts at the relavant local scales and fishing intensites.

Limitations in the findings of this study include the lack of historical lugworm size or abundance data to observe the changes over time from fishing pressure. Comparative and experimental studies were designed to infer impacts, but variability between sites, and scale dependance of impacts limits the ability of these methods to accuratley observe lasting fisheries impacts. Anecdotal reports have suggested reduced density at Boulmer over time, with one collector stating that "it takes twice as long to collect half the worms" (personal communication with collectors). In the absence of historical lugworm data, these claims cannot be investigated further unless ongoing monitoring data is established to observe ongoing changes.

A further limitation within the experimental study was the inefficency of the exclusion nets to exlcude all lugworms from plots. The exclusion treatment was designed to remove the majority, if not all, lugworms. The nets should have removed the ability of lugworms to maintain a burrow (Volkenborn and Reise, 2007), however this was not the case, with many lugworms remaining within the plots. The exclusion nets stayed in place throughout the treatments, remaining in the original positions upon removal after several months, therefore net movement is not responsible for the method failure. The method of a 1mm mesh inserted at a depth of 10cm has effectively excluded lugworms in previous studies (Volkenborn and Reise, 2006; Volkenborn *et al.*, 2007a; Volkenborn and Reise, 2007; O'Brien *et al.*, 2009; Kuhnert *et al.*, 2010a; Lei *et al.*, 2010), all of which were carried out on the island of Sylt, in the Wadden Sea. The size

of lugworms could be responsible for the failure of this method here; lugworms can reach a mass of 30g in some locations (Schroer *et al.*, 2011), but only 10g maximum in the North of England (Fowler, 1999), with a mean mass of 6g recorded within the BNNC EMS (Chapter 4). This smaller size may mean that lugworms in this study area are capable of maintaining a shallower burrow depth above the exclusion net. Shallower exclusion depths of 5cm and 7cm have been used in other studies (Van Wesenbeeck *et al.*, 2007; Rossi *et al.*, 2013), which may have proved more successful in this study. The unintended smaller reduction in lugworm density (by 35%) with exclusion nets in fact emulated a more realistic fishery impact, as lugworm populations are more likely to be reduced than locally extinct with overexploitation (Van den Heiligenberg, 1987). A slightly reduced lugworm abundance without disturbance has not been achieved in previous studies, as such this study is the first to investigate the effects of a marginally lower lugworm density on the associated community (discussed in section 2.4.2).

Overall, lugworm harvesting at current intensities within the EMS is not resulting in long term discernible impacts on the target species over natural variability. Short term impacts appear to stabilise over longer time scales and larger spatial scales, with larval recruitment capable of maintaining exploited populations at harvestable levels for many decades. There is no direct evidence of declining lugworm populations, and as such lugworm harvesting within the BNNC EMS appears to not significantly impact upon the target species currently.

4.4.2 Impacts upon the sediment community

Negative impacts upon the meso- and macrofaunal sediment communities were evident in both the comparative and experimental studies. Substantially lower infaunal abundance with the presence of digging was the most significant finding, along with signs of reduced species richness, and altered community structure. No negative impacts upon the diversity were observed in either study.

Within the comparative study, Boulmer, being the muddier site, with higher organic content, would be expected to contain a more abundant and diverse infaunal community without disturbance, but be less resilient to disturbance than communities in more mobile sand conditions (e.g. MacDonald *et al.*, 1996; Schratzberger and Warwick, 1998; Ferns *et al.*, 2000; Kaiser *et al.*, 2006a; Roberts *et al.*, 2010).

Boulmer contained the lowest infaunal abundance, less than half those of Newton and Holy Island, suggesting that disturbance has reduced the community. A number of different disturbances could lead to the differences observed between sites, with the heterogeneity of infaunal community structure along a coastline well documented (e.g. Morrisey *et al.*, 1992; Norén and Lindegarth, 2005). Natural variation from habitat (e.g. Thorson, 1950; Gray, 1974; Beukema, 1976; Holland and Dean, 1977; Probert, 1984; Elliot *et al.*, 1998; Ysebaert *et al.*, 2002) and environmental conditions (e.g. Levin *et al.*, 2003; Van Hoey *et al.*, 2004; Green *et al.*, 2014b; Gerwing *et al.*, 2015), anthropogenic impacts such as contamination (e.g. Morris and Keough, 2003; Ruso *et al.*, 2007), or both combined (Mucha *et al.*, 2003; Stark *et al.*, 2005), can influence infaunal communities. Fishing activities can also lead to spatial heterogeneity of sediment communities (e.g. Kaiser *et al.*, 2001; Kaiser *et al.*, 2006a), with bait digging suggested as the cause of Boulmer's low meso- and macrofaunal abundance within this study. This assumption is further supported by both the experimental digging study results (reducing infaunal abundance with increasing digging intensity), and the existing bait digging literature (e.g. Van den Heiligenberg, 1987; Beukema, 1995; Brown and Wilson, 1997). For example, Van den Heiligenberg (1987) found hand digging removed 1.9g of non-target benthic animals from the sediment for every 1g of lugworm harvested, reducing the infaunal biomass by 40%.

Important wading bird prey species were present in relatively low abundances at all sites compared to smaller species. This may be a result of the small core size limitations (discussed in the methods section 4.2.1). Holy Island was the site with the highest mean and total abundances of these species, which is also the most important conservation area for birds out of the study sites (SSSI, Ramsar, and SPA), including many waders which feed on the expansive sand and mud flats there (see Chapter 1, Table 1.1 for further detail and designated species lists). However, Boulmer also appears to remain a good feeding site for birds despite a high level of collection activities, with *L. conchilega* most abundant here. In the experimental study, these important prey species appeared to decrease with the presence of digging, indicating that under certain conditions and high digging intensities, bait digging has the ability to alter preferential prey availability for birds, as seen elsewhere (Van den Heiligenberg, 1987; Masero *et al.*, 2008; Bowgen *et al.*, 2015).

Community structure was directly altered in experimental plots, with reduced taxonomic richness, and a shift in dominance from Annelids to Nematodes. Impacts increased with digging intensity. Communities were also significantly different in the comparative study, but it is less clear how much is due to digging disturbance versus other environmental differences. Some species are more vulnerable to sediment disturbance than others (e.g. Jackson and James, 1979; Chandrasekara and Frid, 1998), which can result in altered communities as opportunistic species increase, and sensitive species decline (Beukema, 1995; Reise, 2001). Nematodes were among the few taxa which did not decline in the experimental plots, similar to the findings of other studies (Watson *et al.*, 2017b). Nematodes are thought to be more resilient to physical disturbance than larger organisms such as macro- or megafauna because they are less likely to be killed by the disturbance, have a relatively high tolerance to low oxygen levels (e.g. burial conditions), and fast recovery rates (Schmidt-Rhaesa, 2014), culminating in the dominance of this taxa post disturbance. In contrast, *Tubificoides* sp. was significantly reduced from disturbance at the experimental site, and was rare or absent at the two collected comparative sites. *Tubificoides* sp*.* inhabit both muddy and sandy sediments (Genis Trait Handbook, 2015), therefore habitat differences are unlikely to be responsible for the differences observed*.* They have limited mobility and as a result has been referred to as 'vulnerable', especially to sediment deposition (Genis Trait Handbook, 2015), suggesting that digging disturbances are likely responsible for the reduced abundances at Boulmer, Newton, and the experimental site, with similar negative impacts also observed for *Tubificoides benedii* from bait digging in midshore areas of the Solent (Watson *et al.*, 2017b).

The reduction of lugworm density by 35% (rather than total exclusion) revealed that even marginally reduced lugworm populations can have significant detrimental impacts on the associated sediment community, with lower taxonomic richness observed in exclusion plots. Lugworms are habitat engineers, altering the state of the habitat, affecting other infaunal species (Lawton, 1994; Wright and Jones, 2006; Volkenborn *et al.*, 2007a; Volkenborn and Reise, 2007; Passarelli *et al.*, 2014). They rework the sediment (Retraubun *et al.*, 1996b; Passarelli *et al.*, 2014), mixing the upper layer (Cadée, 1976; Retraubun *et al.*, 1996b; Risgard & Banta, 1998 as cited by Valdemarsen *et al.*, 2011), in turn destabilising the sediment (Woodin, 1985; Brey,

1991; Flach, 1992), whilst their burrows transport particles and oxygen through the sediment (Reise, 2002), forming unique microhabitats (Banta *et al.*, 1999; Nielsen *et al.*, 2003), and aerating the sediment for other infaunal species (Baumfalk, 1979; Retraubun *et al.*, 1996a; Schroer *et al.*, 2011). Lugworms have both positive and negative impacts upon different species, playing an important role in structuring benthic communities (Brey, 1991; Petrowski *et al.*, 2016). Removing lugworms, and their bioengineering effects, from a shore via bait digging (or experimental exclusion) can result in substantial indirect impacts (Cryer *et al.*, 1987). This has been seen in many lugworm exclusion experiments, with different species effected in various ways, both positively or negatively (Volkenborn and Reise, 2007; Petrowski *et al.*, 2016; Whitton *et al.*, 2016; Sousa *et al.*, 2017). This study is the first to demonstrate that even slightly reduced lugworm abundance (a much more realistic scenario from lugworm overexploitation) can have detrimental community scale impacts.

Currently it appears that lugworm abundance is not reduced within the BNNC EMS in the long term (see section 2.4.1) and therefore these indirect effects are not a priority concern for management at this time. The direct habitat disturbance impacts should be the main concern in conservation plans for lugworm collection. Bait digging disrupts the sediment layering, releases toxins and pollutants (Howell, 1985; Fowler, 1999), reduces organic matter (Watson *et al.*, 2017b), and directly damages and kills infauna (Chandrasekara and Frid, 1998). Bait pumping creates substantially less sediment disturbance during the collection of *A. defodiens*, with much smaller amounts of disturbed sediment and no spoil heaps produced (Fowler, 1999). It appears that bait pumping in Northumberland would have a lower level of impact upon infaunal communities if both methods removed the same number of lugworms. Promotion of this collection method over digging has been considered, but overall seems unsuitable due to the fact that bait pumping can only target *A. defodiens* (Fowler, 1999), which is much scarcer within the BNNC EMS than *A. marina* (see species distribution maps in Chapter 5), providing much lower target stocks and a higher chance of overexploitation.

The lowered infaunal abundance or biomass observed in this study can result in reduced benthic food supply for birds, and the altered community structure could cause food shortages for species with strong prey preferences (Van den Heiligenberg, 1987; Masero *et al.*, 2008; Bowgen *et al.*, 2015), forcing birds to switch

to other prey types or use alternate feeding areas (Beukema *et al.*, 1993). Migratory birds are especially vulnerable to prey decline, relying on a few specific coastal areas during their journey (Skagen and Knopf, 1993; Masero *et al.*, 2008). The BNNC EMS contains multiple SPA designations which are key sites for the protection of important bird populations (NCAONB, 2009), as such activities which may hinder bird populations should be minimised. In the scale of the BNNC EMS, lugworm collection occurs over a small area, leaving vast areas of sediment with natural infaunal biomass for successful bird feeding. None of the highest intensity lugworm collection sites (see Chapter 4 and 5) are located within SPAs currently, and as such effects on birds from reduced prey may not be a major concern for managers.

The changes in the infaunal communities observed from bait digging in this study could also be altering the functional diversity of communities, with the ability to modify ecosystem functioning (Dıaz and Cabido, 2001; Solan *et al.*, 2004; Tillin *et al.*, 2006). Species within a community play various roles, with contrasting interactions and processes. A high diversity of functional traits has been shown to maintain ecosystem processes (Dıaz and Cabido, 2001), with extinctions predicted to reduce bioturbation in marine benthos (Solan *et al.*, 2004). Further study into the functional trait effects of bait digging is needed to see how the altered communities observed here may have wider reaching consequences on the ecosystem.

Overall, there are negative impacts occurring, but whether they are significant at the larger EMS ecosystem scale remains unknown. Further study on wider ecosystem effects (other than birds) of observed community impacts (reduced abundance, altered community structure, and functional diversity) are needed to fully inform management decisions.

4.4.3 Recovery of the target species and associated sediment communities after digging disturbance

Both the target species and infaunal communities recovered well after the experimental disturbances ended, but it is important to consider whether the high recoverability observed translates into larger scales both spatially and temporally within the BNNC EMS.

Recovery rate of infaunal communities is dependent on many factors, such as season (Zajac and Whitlatch, 1982a; Zajac and Whitlatch, 1982b; Alongi, 1990; Ford *et al.*, 1999), scale of disturbance (Reise, 2001), sediment characteristics (Dernie *et al.*, 2003), structure of the original community (Jackson and James, 1979; Beukema, 1995; Fowler, 1999; Watson *et al.*, 2007), and the method of recolonization (i.e. migration or recruitment) (Reise, 2001). As such, there is high variability in the recovery rates observed between previous bait digging studies, ranging from one month to 3 years for the target species (Blake, 1979a; Cryer *et al.*, 1987; Beukema, 1995), and 140 days to 5 years for infaunal communities (Van den Heiligenberg, 1987; Beukema, 1995).

This study's findings suggesting full infaunal recovery within 77 days is very fast, likely due to a combination of various beneficial artefacts of the study site and experimental design. The timing for the recovery period (during summer) may have accelerated the recovery rate, as recolonization of infauna is usually faster in the spring and summer (Zajac and Whitlatch, 1982a; Zajac and Whitlatch, 1982b; Alongi, 1990; Ford *et al.*, 1999). Recovery from current collection within the BNNC EMS may be slower after digging intensity reduces in late winter/early spring. Additionally some insensitively collected sites (e.g. Boulmer) experience bait digging almost year round (see Chapter 4 for details), with very little 'undisturbed' time for recovery to take place. The experimental plots were also small scale disturbance compared to bait digging activity, surrounded by undisturbed refuge areas, allowing for maximum migration of infauna into previously disturbed plots, rather than having to rely on planktonic larvae or post larval drifters to recolonize plots on a larger scale (Reise *et al.*, 2001). The short time scale of the experimental study does not allow for the examination of long term cumulative impacts, and the effects on subsequent recovery (Brown and Wilson, 1997). The macrofaunal community at Fenham Flats, the experimental site, did not have a high proportion of sensitive species such as large bivalves or burrowing echinoderms, which would be expected to recover more slowly (Jackson and James, 1979; Beukema, 1995; Fowler, 1999; Watson *et al.*, 2007).

Harvested sites around the BNNC EMS vary widely in many of the aspects discussed above, and as such the recovery rate observed in this experimental study is very unlikely to relate to those elsewhere. Recovery after large scale, ongoing,

disturbances from fishers is likely to be slower than that observed at Fenham Flats. However, the experimental study shows that recovery is likely under the right conditions, and suggests that no-take periods on collected shores may be adequate to allow full recovery of the infaunal communities.

4.5 Conclusions

This chapter presents evidence which provides baseline information to help inform management plans for the BNNC EMS and other protected areas within the UK. The use of both comparative and experimental studies combined provides two separate evidence bases which can be compared to support individual findings, elucidating potential (short-term) and actual (long-term) impacts of lugworm collection activities within the BNNC EMS.

Lugworms play an important role in intertidal communities (Lawton, 1994; Wright and Jones, 2006; Volkenborn *et al.*, 2007a; Volkenborn and Reise, 2007) and are an important prey species to both birds and fish (Evans *et al.*, 1979; Pocklington and Wells, 1992). Results from this research suggest that impacts on lugworms are not discernible against natural background variability, and that at current, local collection levels, lugworm collection appears be having no impact at the target species level, with stable lugworm communities throughout the study area. However, cumulative impacts over longer timescales may change this, especially if harvesting intensity increases in the future.

Bait digging in Northumberland is however causing substantial negative impacts at the infaunal community level, which is important because infaunal communities are a key sub-feature of the BNNC SAC (European Union Council Directive. 92/43EEC, 1992), and as such should be maintained/protected. The effects of these reduced and altered infaunal communities and the role that plays in the ecosystem and overall site integrity needs to be explored further.

Chapter 5 : Investigation of the Impacts of Periwinkle Collection within the Berwickshire and North Northumberland Coast European Marine Site

5.1 Introduction and Rational

Fisheries impacts are well studied globally (e.g. Dayton *et al.*, 1995; Auster *et al.*, 1996; Thrush *et al.*, 1998; Turner *et al.*, 1999; Collie *et al.*, 2000; Coleman and Williams, 2002; Kaiser *et al.*, 2006b; Williams *et al.*, 2008; Smith *et al.*, 2011), with the investigation of intertidal fisheries gaining more traction in recent years (e.g. Kaiser *et al.*, 2001; Thompson *et al.*, 2002; Berthelon *et al.*, 2004; Masero *et al.*, 2008; Sheehan *et al.*, 2010; Erlandson *et al.*, 2011; Crossthwaite, 2012; Bertocci *et al.*, 2014; Clarke and Tully, 2014; Manríquez *et al.*, 2016). However, evidence on the potential effects of fisheries on protected species and habitats is still lacking in many areas, which is why UK conservation authorities (Natural England, Defra, IFCAs, the MMO, Cefas, and JNCC) are looking to gather additional evidence to determine where and why management is needed (Moffat, 2015). The revised approach to commercial fisheries management in EMSs (the main driver of the push for increased evidence) is currently being implemented, with the aim of producing well-managed fisheries (MMO, 2014b; Moffat, 2015). By June 2014, seventeen MMO or IFCA byelaws were in place to protect sensitive features (Moffat, 2015), yet many fisheries-feature interactions remain unassessed. One of the key areas identified as requiring additional evidence is the impacts of hand gathering on intertidal rocky reefs (MMO, 2014b).

Hand gathering impacts on rocky shores have been studied around the world (e.g. Kingsford *et al.*, 1991; Keough *et al.*, 1993a; Fanelli *et al.*, 1994; Siegfried *et al.*, 1994; Fletcher and Frid, 1996; Lindberg *et al.*, 1998; Sharpe and Keough, 1998; Murray *et al.*, 1999; Quigley, 1999), with an emphasis on areas with well-established reserves and no-take zones, such as those of Australia and South-Africa (Underwood and Kennelly, 1990; Keough and King, 1991; Kingsford *et al.*, 1991; Keough *et al.*, 1993a; Underwood, 1993; Siegfried *et al.*, 1994; Sharpe and Keough, 1998; Thompson *et al.*, 2002). Within Europe, the target species of multiple studies is the periwinkle, *Littorina littorea* (e.g. Quigley, 1999; Berthelon *et al.*, 2004; Crossthwaite, 2012), likely due to their widespread distribution (Jackson, 2008b) and popularity of collection (Fowler, 1999; Cummins *et al.*, 2002). Yet uncertainties remain, especially in relation to local, measured, and realistic collection intensities.

Rocky shore exploitation can have major impacts upon the target species (reveiwed by Thompson *et al.*, 2002). Abundance reductions from harvesting are commonly

observed (Quigley, 1999; Berthelon *et al.*, 2004). However, alterations are cumulative, and do not appear in the short term, with historical collection playing a large role (Crossthwaite, 2012). Harvesting often targets the largest individuals of a population, resulting in altered size structures (e.g. Castilla and Duran, 1985; Lindberg *et al.*, 1998; Thompson *et al.*, 2002; Roy *et al.*, 2003; Berthelon *et al.*, 2004), with mean or modal sizes reduced by 10-20% in most studies (summarised in Keough *et al.*, 1993a).

Removing organisms from a shore can alter community interactions (Berthelon *et al.*, 2004), which in turn can modify community structure. Removal of *L. littorea* has the potential to reduce grazing pressure, increase algal cover, enhance sedimentation, and control the recruitment of sessile organisms (e.g. Petraitis, 1989; Cervin and Aberg, 1997; Buschbaum, 2000; Crossthwaite, 2012). Physical disturbance of the habitat and organisms can also have negative effects, from trampling and stone-turning (Fowler, 1999; Berthelon *et al.*, 2004; Tyler-Walters and Arnold, 2008; JNCC and Natural England, 2011).

Comparative and experimental methodologies are used to investigate fishing impacts throughout the scientific literature (FAO, 2005). Within comparative studies, the community state indicates the impacts, whilst before and after measurements are used in experimental methods (FAO, 2005; Hughes *et al.*, 2014). Comparative studies are commonly used to investigate rocky shore impacts from fishing – often comparing communities inside and outside of protected areas with long standing no-take zones (e.g. Keough and King, 1991; Keough *et al.*, 1993a; reveiwed in Thompson *et al.*, 2002). Periwinkle collection impacts have mostly been studied with experimental methodologies to date, predominantly exclusion cages (e.g. Petraitis, 1989; Cervin and Aberg, 1997; Quigley, 1999; Buschbaum, 2000; Hancock and Petraitis, 2001; Cervin *et al.*, 2004), with few studies using comparative methods (Quigley, 1999; Berthelon *et al.*, 2004). Using exclusion experiments to examine the indirect effects of periwinkle collection on a community (rather than simply exploring community interactions) assumes that periwinkle fisheries are reducing the target species stocks, which may not be the case in some areas (Berthelon *et al.*, 2004). The conditions of experimental studies need to be representative of actual fishing impacts for the findings to be useful to conservation bodies and policy makers.

The aim of this chapter is to investigate differences in the population size and structure of the target species, *L. littorea,* and how any observed differences relate to the gradient of fishing pressure within the BNNC EMS. Associated macroalgal and macroinvertebrate assemblages are described, and comparative methods used to explore observable impacts from actual fishing pressure, notwithstanding the limitations of such an approach. The decision to forego experimental methodologies is discussed.

5.2 Methods

5.2.1 Comparative study

Site Selection

Three shores were required for comparison, each with a different level of collection pressure: no collection, low collection, and high collection. Shores within the BNNC EMS with appropriate collection pressures were identified on the basis of preliminary shore visits combined with advice from expert authorities (Natural England and the Northumberland Inshore Fisheries Conservation Authority) to establish known periwinkle harvesting activity. The selected shores were observed regularly from December 2013 to July 2014. Each site was visited at low tide 1-2 times per month throughout the monitoring period to estimate the intensity of periwinkle collection occurring at each, validating the assumed collection pressure classifications. The observations were made on a mix of both weekdays and weekends, and under various environmental conditions (e.g. weather and seasons), to remove confounding effects presumed to influence harvesting behaviour (Fowler, 1999; Cummins *et al.*, 2002). At each visit, the number of periwinkle collectors present at each site was recorded.

Marshall Meadows Bay (O.S. Grid Reference NT982568) was selected as the 'no collection' site, being a remote and difficult to access shore (single access route down the 50ft cliffs via a disused subterranean tunnel constructed in the 1800s, with a concealed entrance located on private gated land). A rocky stretch on the south-west corner of Holy Island (O.S. Grid Reference NU124416) was chosen as the 'low collection' shore, due to anecdotal collection despite the remote location. Boulmer rocky shore (O.S. Grid Reference NU270148) was selected as the 'high collection' shore, with considerable collection observed in preliminary visits.

The locations of each site in relation to the position within the BNNC EMS are shown in Figure 5:1.
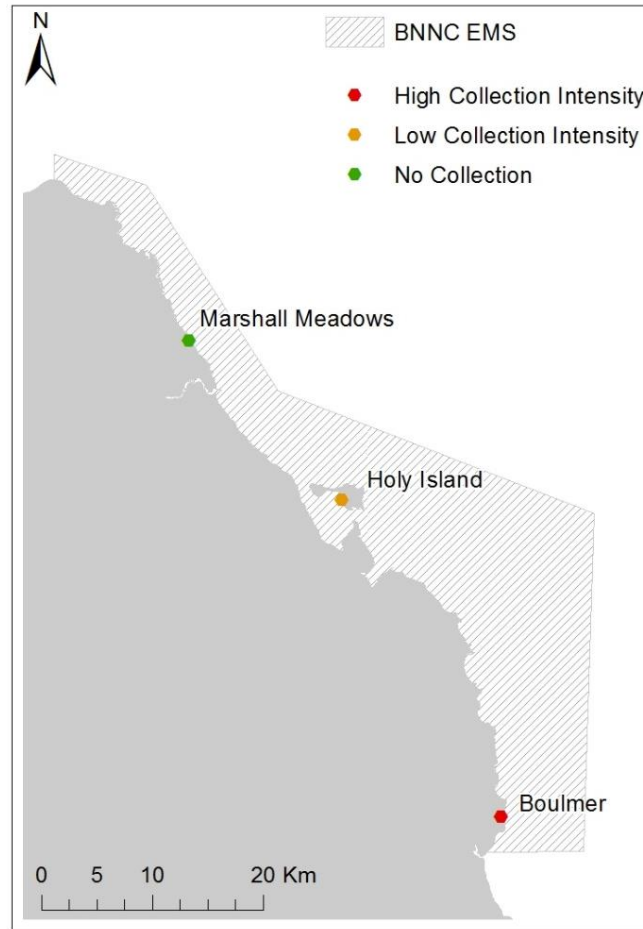
Figure 5:1: Locations of sample sites: Boulmer (high collection pressure), Holy Island (low collection pressure), and Marshall Meadows (no collection), within the BNNC EMS.
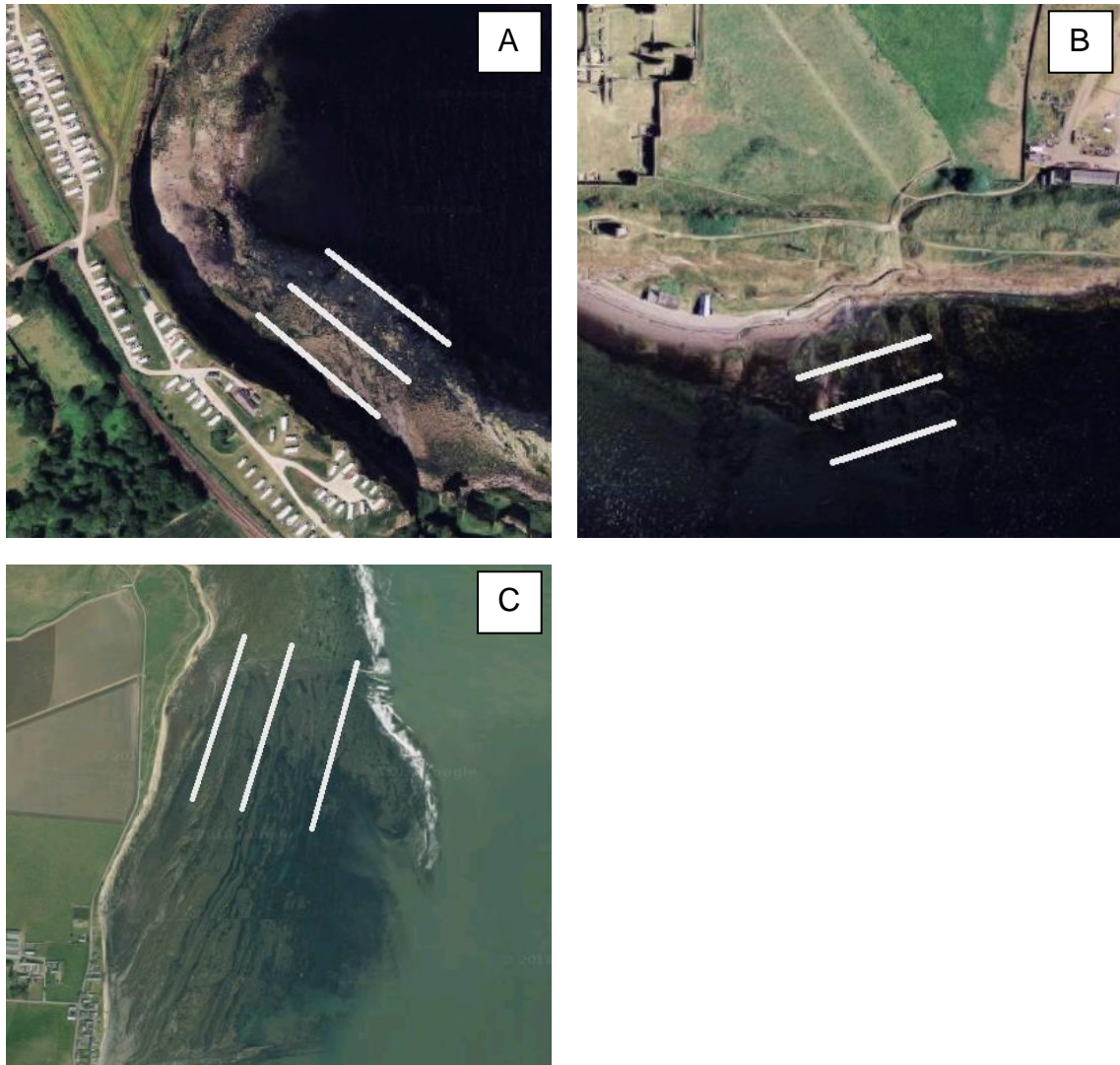
Figure 5:2: Aerial images of each study site (Map data @2018 Google). A = Marshall Meadows, B = Holy Island, C = Boulmer. Approximate transect locations for high, mid, and low shore are displayed by light grey lines.

Sampling

Sampling was carried out in March 2014, at low spring tides. Preliminary observations revealed periwinkle collection occurs at any shore height at which periwinkles are present. At each shore, ten quadrats (50 x 50 cm) were placed randomly along 3 shore height transects at the lowest, middle, and highest levels of the zone where periwinkles are found locally (the heights of which can be seen in Table 5:1). Within each quadrat, all *L. littorea* were counted and shell height measured using Vernier callipers.

Table 5:1: Shore heights (meters above chart datum) of low, mid, and high vertical zones of periwinkle distribution, at the three rocky shore study sites.

Shore	Low	Mid	High
Boulmer	0.88 m	2.88 m	4.88 m
Holy Island	0.72 m	3.82 m	1.82 m
Marshall Meadows	0.80 m	1.80 m	2.70 m

Within the same thirty quadrats, abundance of all other macroalgae and macroinvertebrate taxa were recorded to species level where possible. Count data was recorded for most fauna, whilst percentage cover was used for seaweeds, sessile organisms, and encrustations.

5.2.3 Experimental study

Experimental methods within the scientific literature were explored and assessed for suitability within Northumberland, including time and budget constraints. The logistics and requirements for cage exclusion/manipulation experiments are displayed and discussed. Requirements for experimental set-up were gained from a variety of studies (for e.g. average replicates completed, average run time), and instructions for manufacturing cages from Miller (2006). Costing for equipment was taken from McMaster-Carr Supply Company website (www.mcmaster.com), as recommended for sourcing cage supplies by Miller (2006) (dollars were converted to pounds using the current exchange rate at time of viewing). SWOT (strengths, weaknesses, opportunities, and threats) analysis was also conducted, to consider important factors of the study parameters, and highlight pros and cons of conducting the proposed experiments.

Ultimately, it was decided that experimental study was unsuitable and unnecessary for this study site under the constraints. The evidence used to reach this decision is presented in the results section.

5.2.4 Data Analysis

Univariate statistics were analysed using Minitab version 17, and multivariate with Primer software. Differences between sites were tested using ANOVA where parametric assumptions were met (normal distribution (or normalized using log or square-root transformations) and similar variances). Kruskal-Wallis was used where normality assumptions could not be met (Underwood, 1997; Dytham, 2011). Diversity

was measured using the Shannon Wiener function (H), which was calculated for each sample and averaged for sites. Community structure was analysed using Bray Curtis Similarity (on square root transformed averaged data), with results expressed in Multidimensional scaling (MDS) plots. SIMPER analysis was used to determine the species responsible for the differences observed, which were subsequently plotted graphically.

5.3 Results

5.3.1 Collection Pressure and Habitat characteristics

Observations of periwinkle collection at the comparative sites (Boulmer, Holy Island, and Marshall Meadows) validated the assumptions made from expert advice and preliminary visits. It was confirmed that Boulmer has the highest collection pressure, Holy Island low collection, and Marshall Meadows no collection (Table 4:1) occurring on observed dates.

Table 5:2: Validation of the collection pressure classifications assigned to each shore from observations recording the number of periwinkle collectors present per shore visit (visited regularly between December 2013 and July 2014). Averages of collectors presented as means with standard deviation. Boulmer n=13, Holy Island n=6, Marshall Meadows n=4.

Location	Collection Pressure	Average no. collectors per visit	S.D
Boulmer	High	1.38	1.04
Holy Island	Low	0.17	0.41
Marshall Meadows	Not Collected	0	0

All three shores are moderately exposed, and dominated by boulders and bedrock, resulting in largely similar habitat characteristics. However, Holy Island was the most dissimilar site, with an unusual shore gradient (highest at mid shore), and the presence of shingle and pebbles at the high shore.

5.3.2 Comparisons between sites with differing collection pressures
Target species

The median densities of *Littorina littorea* per quadrat ($0.25m^2$) are significantly different between sites (Kruskal-Wallis, H = 17.75, df = 2, P < 0.001). The lowest average density was recorded at Marshall Meadows, the uncollected site (median = 1 ± 52 range). The median densities for all sites can be seen in Figure 4:9. Figure 4:10 shows the median periwinkle shell heights at each site, which are significantly different between sites (Kruskal-Wallis, H = 113.01, df = 2, P < 0.001). The largest average periwinkle shell height was observed at Marshall Meadows, the uncollected shore (median = 26 ± 28 range), which also had the largest proportion of large periwinkles (over 30mm). The maximum shell height was 33mm, which was recorded at both Marshall Meadows and Boulmer.
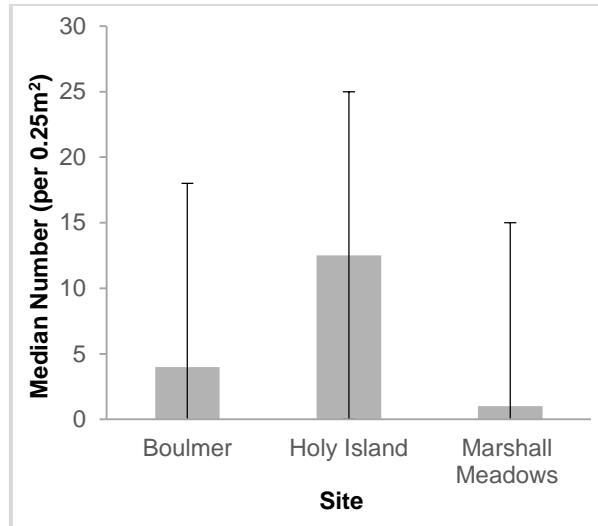
Figure 5:3: Median (± range) number of periwinkles per 0.25m² from three sites of varying collection pressure (Boulmer = high collection pressure, Holy Island = low collection pressure, Marshall Meadows = no collection), sampled March 2014; n = 30 for all sites.
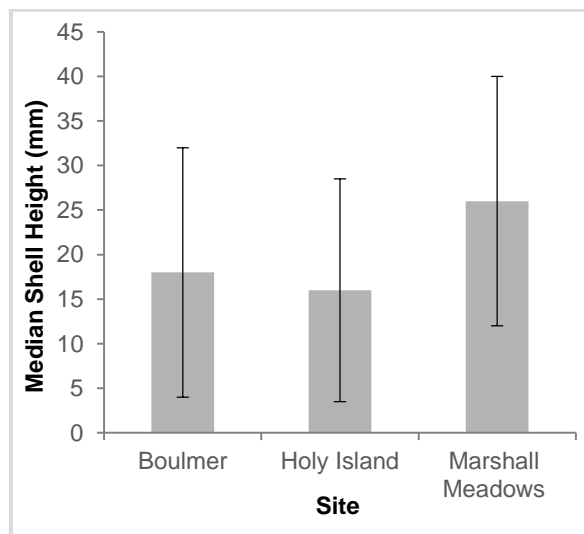
Figure 5:4: Median (± range) shell heights (mm) of periwinkles from three sites of varying collection pressure (Boulmer = high collection pressure, Holy Island = low collection pressure, Marshall Meadows = no collection), sampled March 2014, with all shells measured from each quadrat (30 per site). Boulmer n = 280, Holy Island n = 714, Marshall Meadows n = 127.

Rocky Shore Community

The occurrence and abundances of taxa recorded at each site for all quadrats combined can be seen in Table 5:3. Crustacea and Mollusca dominate the faunal communities' at all three sites, whilst algal taxonomic richness is highest at Boulmer and Marshall Meadows. Overall, the communities present at Boulmer and Marshall Meadows appear more similar than those at Holy Island (Table 5:3). The three most abundant faunal community species (excluding periwinkles) were *Patella vulgata, Gibbula cineraria*, and *Pagurus bernhardus*. The mean (+/- SD) for each of these species at each site can be seen in Figure 5:5. Holy Island was the most distinct site, with no *P. bernhardus* present, and much lower abundances of *G. cineraria* compared to Marshall Meadows and Boulmer.

The average taxonomic richness is significantly different between shores (Kruskal-Wallis, H = 30.88, df = 2, P < 0.001), with the highest richness recorded at Marshall Meadows (median = 8 ± 11 range), and the lowest at Holy Island (median = 4 ± 9 range) (Table 5:4). The average floral and faunal abundances were statistically similar between sites for both percentage cover and individual count taxa (Kruskal-Wallis, H = 4.48, 1.83, df = 2, 2, P > 0.1) (Table 5:4). Diversity (average Shannon's diversity) was statistically different between shores (ANOVA, F = 15.45, df = 2, 89, P < 0.001), with the highest recorded at Marshall Meadows (mean = 1.53 ± 0.31 SD), and lowest at Holy Island (mean = 0.94 ± 0.42 SD) (Table 5:4).

Table 5:3: Total abundance of count data for faunal taxa and presence (+) and absence (left blank) of taxa recorded as percentage cover, within quadrats (50x50cm) collected from three shores of differing collection pressures (Boulmer = high collection pressure, Holy Island = low collection pressure, Marshall Meadows = no collection). Samples were collected in March 2014 on low water spring tides (n=30 for all shores).

Species/Taxa	Boulmer	Holy Island	Marshall Meadows
ALGAE			
Ahnfeltia plicata	+		
Ascophyllum nodosum	+	+	+
Cladophora rupestris	+		+
Corallina officinalis	+		+
Fucus serratus	+	+	+
Fucus spiralis	+		
Fucus vesiculosus		+	
Laminaria digitata			+
Mastocarpus stellatus	+		+
Palmaria palmata			+
Ulva intestinalis	+		+
Ulva lactuca	+		+
Unidentified encrusting coralline	+		+
Unidentified red turf	+	+	+
CRUSTACEA			
Cancer pagurus	4	0	5
Carcinus maenas	0	1	1
Galathea squamifera	4	1	5
Pagurus bernhardus	79	0	69
Porcellana platycheles	1	0	0
Semibalanus balanoides	+	+	+
Unidentified isopod	2	1	3
PORIFERA			
Unidentified sponge		+	+
ANNELIDA			
Polynoidae sp.	2	0	1
Pomatoceros sp.	3	0	2
Spirorbis spirorbis	+	+	+
MOLLUSCA			
Anomia ephippium	15	25	0
Gibbula cineraria	71	2	138
Lepidochitona cinerea	5	0	1
Littorina littorea	284	714	130
Littorina obtusata	48	21	11
Mytilus edulis		+	
Nucella lapillus	31	1	21
Patella vulgata	84	58	111
Unidentified nudibranch	0	1	0
ECHINODERMATA			
Asterias rubens	0	1	0
Henricia oculata	1	0	1
Ophiothrix fragilis	4	0	0
CNIDARIA			
Actinia equina	3	0	12
ASCIDIACEA			
Botryllus schlosseri	0	0	1
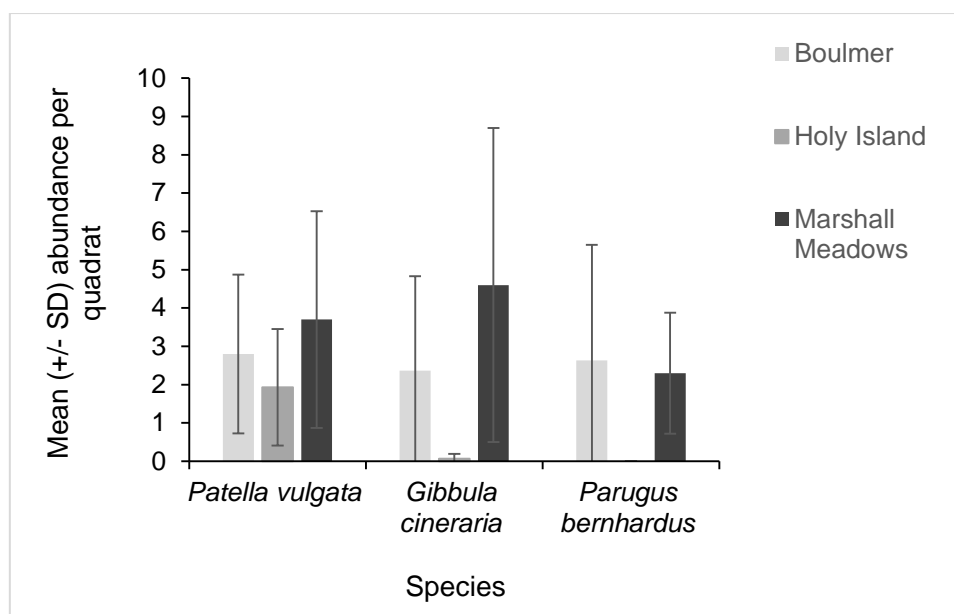
Figure 5:5: Mean (+/- SD) abundances the three most abundant faunal species present at each study site (Boulmer, Holy Island, and Marshall Meadows). n = 30 for all sites.

Table 5:4: Median (± range) faunal abundance (count data only) and taxonomic richness, and mean (± SD) Shannon's diversity for each site with differing collection pressures (Boulmer = high collection pressure, Holy Island = low collection pressure, Marshall Meadows = no collection), sampled March 2014 (n=30).

	Boulmer	**Holy Island**	**Marshall Meadows**
Abundance	14.5 (± 72.0)	22.5 (± 81.0)	10.5 (± 79.0)
Taxonomic richness	7.0 (± 16.0)	4.0 (± 9.0)	8.0 (± 11.0)
Diversity	1.22 (± 0.05)	0.94 (± 0.42)	1.53 (± 0.31)

The community structure of the rocky shore organisisms between sites is significantly different (ANOSIM: Global R=0.312, p=0.1%). Bray Curtis similarity shows that Boulmer and Marshall Meadows have a higher similarity level of around 70%, whilst Holy Island is the most distinct community, with only 45% similarity to either site. The Multi-Dimensional Scaling (MDS) plot of the Bray Curtis similarity (Figure 4:12) for the rocky shore communities showed some discrimination between sites, with a lot of overlap occurring. The similarity grouping overlays at 20 and 30% show that samples from different sites are often more similar than those from within a single sample site; the similarity groupings do not clearly distinguish between sites. SIMPER analysis shows that the main faunal species (greatest % contribution) responsible for the significant differences observed in community structure between the three sites (excluding *Littorina littorea*) are: *Mytilus edulis, Patella vulgata,* and

Gibbula cineraria, which are also some of the most dominant species recorded. The main floral species responsible are: *Fucus serratus, Corallina officinalis, and* unidentified encrusting coralline*.* The total abundances of each faunal species from the SIMPER analysis is displayed in Figure 4:13. *Mytilus edulis* was only present at Holy Island, whilst *Patella vulgata* and *Gibbula cineraria* were present at all sites, both being most abundant at Marshall Meadows.
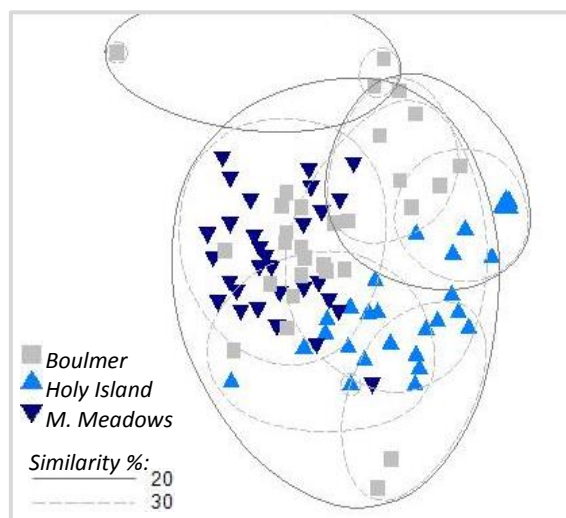
Figure 5:6: Non-metric multidimensional scaling (MDS) ordination of the Bray Curtis similarity based on square root-transformed averaged abundance data of the rocky shore community from sites with differing collection pressures (Boulmer = high collection pressure, Holy Island = low collection pressure, Marshall Meadows = no collection), sampled March 2014. 2D Stress: 0.22. Overlays of Bray Curtis similarity groupings at 25 and 40%.
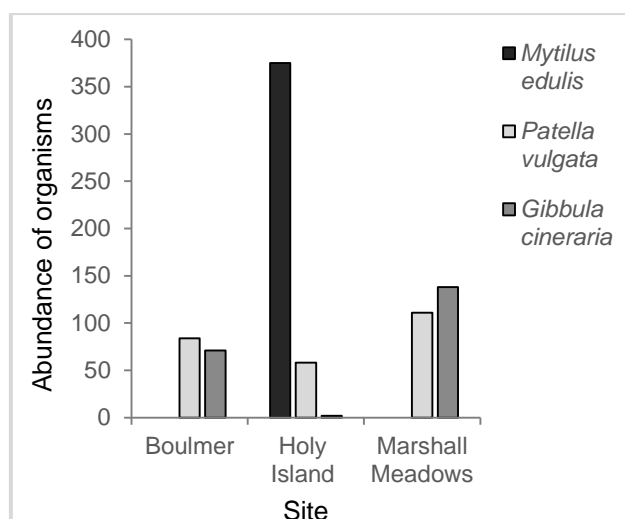
Figure 5:7: Total abundances of the three species most responsible for the difference in community structure per site (Boulmer = high collection pressure, Holy Island = low collection pressure, Marshall Meadows = no collection), sampled March 2014 (n = 30 quadrats per site = 7.5m^2).

5.3.4 Experimental study

Cage exclusion or manipulation experiments were considered for use in this study, to provide empirical evidence on the effects of varying *Littorina littorea* density and grazing pressure on the associated community, evaluating the indirect effects of harvesting. Grazer exclusion experiments have previously been used to elucidate many grazer interactions and their influences on communities (e.g. Menge and Lubchenco, 1981; Menge *et al.*, 1985; Petraitis, 1989; Geller, 1991; Williams, 1993; Williams, 1994; Cervin and Aberg, 1997; Buschbaum, 2000; Fong *et al.*, 2000; Hancock and Petraitis, 2001; Bazterrica *et al.*, 2007; Scheibling *et al.*, 2008; Perez *et al.*, 2009; Mrowicki *et al.*, 2014; Guerry and Menge, 2017), the vast majority of which use cages to alter the natural abundances of grazers. Using cages to manipulate periwinkle density and/or size could be used to infer the response of local rocky shore communities to altered periwinkle stocks from unsustainable harvesting. The logistics and requirements in terms of equipment, time, and cost were considered for this type of experimental set-up, the outcomes of which can be seen in Table 5:5 and Table 5:6. A total time estimate of 35 days is conservative given the need to monitor, replace, and repair storm damaged or vandalised cages.

Table 5:5: Considerations and requirements for a cage exclusion/manipulation experiment to investigate the impacts of periwinkle harvesting on rocky shore communities.

Considerations	Requirements
Treatments	6 treatments: · Natural Density, Natural Size · Natural Density, Reduced Size · Reduced Density, Natural Size · Reduced Density, Reduced Size · Cage Control · Open Control
Replicates	6 replicates per treatment
Total number of plots	36 plots (30 cages or partial cages, 6 open)
Experimental run time	12 months
Equipment for set-up	· Stainless steel woven wire mesh · Vise-grips · Tin snips · C-clamps · Hammer · Hammer drill (masonry) · Plastic masonry anchors · Stainless steel lag bolts (¼") · Washers
Time for cage production	6 days
Time for installation and set-up	5 days
Time for sampling and maintenance	24 days (2 days per month)
Total time investment	35 days

The equipment requirements for this type of experimental set-up are large, and the run time is long (Table 5:5). An experimental run time of 12 months was chosen as an average based on other cage studies. It can take as long as three years to see full dominance shifts (Menge *et al.*, 1985), but community alterations have also been observed in much shorter timescales (e.g. Petraitis, 1989; Buschbaum, 2000; Hancock and Petraitis, 2001; Scheibling *et al.*, 2008). Around 12 months is a common run time for gastropod inclusion/exclusion studies (e.g. Williams, 1993; Mrowicki *et al.*, 2014; Guerry and Menge, 2017), being enough time to observe changes, and allowing seasonal variability to be considered. Six treatments is deemed adequate to investigate both size and density alterations in periwinkle stocks. Around six replicates of each treatment seems common in previous studies (e.g. Geller, 1991; Williams, 1993; Williams, 1994; Fong *et al.*, 2000; Bazterrica *et al.*, 2007; Scheibling *et al.*, 2008; Guerry and Menge, 2017).

Running a periwinkle manipulative cage study is extremely costly compared to that of lugworm exclusion (Chapter 4). The total estimated cost of £4,277.43 is more than this study can accommodate (Table 5:6).

Table 5:6: Example costs of the equipment, time, and travel required to set up, maintain, and sample cage experiments.

Category	Item Specifics	Price
Equipment for set-up	30 x Corrosion resistant stainless steel woven wire sheets (4x4 mesh, 0.54" wire diameter, 24x24")	£1,368.60
	1 x Vise-grip locking pliers, long nose	£9.97
	1 x Smooth-edge high-force sheet metal cutter (snips)	£23.45
	2 x Iron C-clamp, 6" Maximum - 0" min opening	£40.16
	1 x Hammer for sheet metal forming, 4" head length	£28.31
	1x Cordless hammer drill, 18 volt	£303.20
	120 x Tri-lobe anchor for concrete, ¼" screw size, 1.5" long	£9.02
	120 x Hex head stainless steel screws, 1¼" long	£20.75
	120 x Stainless steel oversized washer for ¼" screw	£33.35
	Total set-up equipment costs	£1,836.81
Potential cage replacements	5 spare cages for timely replacement if damaged	£238.62
Field assistant	Need two people for all stages – pay 1 field assistant for 35 days (6 hours per day, £8 per hour)	£1,680.00
Travel	Travel distance depends on chosen site, but approx. 120 miles round trip per site visit (29 visits at 15p per mile)	£522.00
Total Costs		£4,277.43

SWOT (strengths, weaknesses, opportunities, and threats) analysis was used to summarise and compare the pros and cons of the cage exclusion/manipulation method for use in this study (Table 5:7). The weaknesses and threats of using this experimental method appear to significantly outweigh the strengths and opportunities (Table 5:7). One major, and overwhelming weakness is that it can only investigate the indirect impacts associated with unsustainable harvesting. If periwinkle collection within the BNNC EMS reduced periwinkle densities or sizes, then this method would elucidate the associated community alterations from these changes. However, there is currently no evidence of these changes in periwinkle stocks. Although it can be useful to know the risks of collection activities if they were to be unsustainable in the future, it is most important for management consideration to know the impacts associated with the current stock state, leading to management which is appropriate for actual harvesting intensities.

Table 5:7: SWOT analysis of the cage exclusion/manipulation experimental method.

	Cage Exclusion/Manipulation Experiments
Strengths	· Can be used show effects of reduced periwinkle abundance · Can be used show effects of reduced periwinkle size
Weaknesses	· Only investigates indirect impacts of harvesting on community · Very costly · Experimental set-up difficult and timely · Maintenance requirements high · Long experimental run time required to observe changes · Difficult to locate a suitable site · Cages can influence effects, by altering water flow, etc.
Opportunities	· Could be used in combination with experiments to test direct impacts, e.g. trampling or boulder turning
Threats	· Cages damaged or removed by people · Cages lost in strong seas

5.4 Discussion

Periwinkle collection at current intensity levels within the BNNC EMS does not appear to be negatively impacting neither the target species nor rocky shore faunal and floral communities. Periwinkle populations are maintained at harvestable levels at highly collected shores, and communities likely vary from natural variation, rather than harvesting effects.

5.4.2 Impacts upon the target species - Periwinkles

Neither periwinkle size nor density appear to be correlated to harvesting pressures at current exploitation levels, with Boulmer, the heavily collected site, having a relatively high density and large sizes. Periwinkle densities can differ considerably between locations (e.g. Janke, 1990; Vadas, 1992; Wilhelmsen and Reise, 1994; Buschbaum, 2000; Carlson *et al.*, 2006). Natural density variation between shores plays a stronger role here than the impact of harvesting, likely due to other factors such as habitat selection (e.g. Moore, 1937; Newell, 1958; Vermeij, 1972; Gendron, 1977; Carlson *et al.*, 2006; Storey *et al.*, 2013). For example, the presence of rock pools and high rugosity are known to appeal to periwinkles (Newell, 1958; Carlson *et al.*, 2006). Periwinkles were most abundant, but smallest, at Holy Island, where a low level of collection occurs. This suggests that recruitment at this location is high, but growth is slow, possibly due to the low availability of ephemeral algae (a key food source (Lubchenco, 1983; Watson and Norton, 1985; Barker and Chapman, 1990; Norton *et al.*, 1990)) compared to the other sites.

If Boulmer, the high collection pressure shore, was being negatively impacted by the current harvesting levels, it would be expected that the periwinkle stock would be reduced (e.g. Quigley, 1999; Roy *et al.*, 2003; Berthelon *et al.*, 2004) and/or overfishing would have resulted in a smaller average size and altered population dynamics as seen in previous harvesting impact studies (e.g. Castilla and Duran, 1985; Lindberg *et al.*, 1998; Jackson and Sala, 2001; Dayton *et al.*, 2002; Thompson *et al.*, 2002; Roy *et al.*, 2003; Berthelon *et al.*, 2004). In both measurements (average density and size), Boulmer had intermediate results; although it is possible that Boulmer had naturally higher density and body size periwinkle populations and has been altered by collection, densities or shell height are not depleted much beyond those of a largely uncollected site. The largest shell height recorded in this

study was 33mm, which was observed at both Marshall Meadows and Boulmer. Previous studies at Boulmer have recorded the largest shell height to be 28mm (Quigley, 1999) and 30mm (Morrell, 1976). This suggests that continued harvesting of periwinkles at Boulmer in the medium term (over the last 50 years) has not lead to a reduction in maximum shell height.

The sustained relatively high densities and shell heights observed at Boulmer despite high collection levels may be due to *L. littorea* ability to recolonise from high dispersive larval recruitment originating from uncollected shores (Berger, 1973; Johannesson, 1988; Jackson, 2008b). Additionally, Boulmer is a very large rocky shore, so refuge populations will remain in areas not frequented by collectors. Smaller, more isolated shores elsewhere in the BNNC EMS may not be as resistant to periwinkle harvesting as those observed in this study.

With the recent craze for 'foraging' (Wright, 2009; Mabey, 2012), there are concerns that small scale recreational collection of periwinkles may increase further in the near future, and as such impacts should be monitored going forward.

5.4.2 Impacts upon the rocky shore community

Previous studies have observed impacts from harvesting occurring at the rocky shore community level, from either physical damage to the habitat such as boulder turning (Morris *et al.*, 2011; Crossthwaite, 2012) and trampling (Brosnan and Crumrine, 1994; Ferreira and Rosso, 2009), or secondary effects of altered target species abundance and size via food web interactions (Castilla *et al.*, 1985; Branch and Moreno, 1994; Cervin and Aberg, 1997; Sharpe and Keough, 1998; Lirman, 2001; Keuskamp, 2004). Since no effect on periwinkle size nor abundance has been recorded at these study sites, any community differences between the sites in this study cannot be due to the secondary effects of hand gathering activities.

The community data from this study does not reveal any patterns to infer a significant negative impact from periwinkle collection. Despite the contrasting harvesting regimes, the communities at Boulmer and Marshall Meadows were highly similar when both species/taxa presence/abundance and community structure were compared. Marshall Meadows had the highest average taxonomic richness, and diversity, however, there is no evidence that this is due to the lack of periwinkle harvesting. Although the community structure differed slightly between shores, it

does not appear that the main species responsible differ due to periwinkle harvesting occurrence.

Large variation in rocky shore communities is common within the study area (Big Sea Survey data, personal communications). The three most abundant faunal species recorded in this study (*P. vulgata, G. cineraria*, and *P. bernhardus*) also varied widely in recorded abundances in the Big Sea Survey data, where many shores were sampled within the North-East of England. For example, in this study, *G. cineraria* was most abundant at Boulmer and Marshall Meadows, with very low abundances recorded at Holy Island. Within the Big Sea Survey data, *G. cineraria* abundance also varied significantly between shores, being absent or present in very low abundance at some sites (e.g. Beadnell, Howick, Craster, Whitburn, Colywell Bay, Seaton Sluice, Seaham Harbour, St Mary's Island, etc.) and numerous at others (e.g. Low Newton, Cresswell, Eyemouth, Alnmouth, Boulmer, Seahouses, Hauxley, etc.). Similar large differences between shores are present for many of the species recorded in the Big Sea Survey, highlighting the large degree of community variation within the region and the BNNC EMS.

A caveat of comparative methodologies for assessing impacts is the presence of variability between sites due to both natural differences and other anthropogenic stressors (Thompson *et al.*, 2002). Rocky shore communities are spatially and temporally heterogeneous, making defining an 'unimpacted' condition challenging, and complicating the detection of change from anthropogenic activities (Hartnoll and Hawkins, 1980).

Natural variation between shores can be due to various environmental factors such as: wave action/exposure (e.g. Bustamante and Branch, 1996; McQuaid and Lindsay, 2007; Blamey and Branch, 2009), biogeography and hydrology (e.g. Menge *et al.*, 2003; Cole and McQuaid, 2010), nutrient supply (e.g. Menge, 2000), climate/temperature (e.g. Menge *et al.*, 2008), and habitat complexity and structure (e.g. Seapy and Littler, 1978; Beck, 2000; Kelaher and Carlos Castilla, 2005; Kostylev *et al.*, 2005). The similarity of all study sites being classified as moderately exposed should minimise observable community differences from wave action. However, there are clear differences in habitat structure between sites, with Holy Island having the most distinct structure of the three study sites (containing pebbles,

and few rock pools, etc.). Similarly, there are likely small differences in the average sea temperatures between sites, with the more northern sites experiencing lower average temperatures (e.g. Berwick August average of just 14.4 degrees Celsius versus Blyth August average of 15.3 (seatemperature.org)). Natural variation needs to be considered in the analysis and interpretation of findings. For example, *Gibbula cineraria* favour seaweed, stones, and rock pools as habitat (Hayward and Ryland, 1995), which can explain the higher occurrence at Marshall Meadows and Boulmer compared to Holy Island.

Additional anthropogenic effects (other than periwinkle harvesting) must also be considered in impact studies. Anthropogenic stressors which have the ability to impact upon rocky shore communities include: trampling (e.g. Povey and Keough, 1991; Brosnan and Crumrine, 1994; Fowler, 1999; Berthelon *et al.*, 2004; Tyler-Walters and Arnold, 2008; Ferreira and Rosso, 2009; JNCC and Natural England, 2011), mining (Pulfrich *et al.*, 2003a; Pulfrich *et al.*, 2003b), eutrophication (e.g. Kraufvelin *et al.*, 2006; Worm and Lotze, 2006; Arévalo *et al.*, 2007; Kraufvelin, 2007), and of course intertidal harvesting (e.g. Sharpe and Keough, 1998; Moreno, 2001; Berthelon *et al.*, 2004; Davenport and Davenport, 2006). A key difference between the study sites is the amount of trampling and disturbance to the sites on a regular basis. Holy Island rocky shore experiences high foot traffic despite having a low periwinkle collection pressure (personal observation). It is located on a tourism hotspot, which is very popular with walkers and sightseers, and therefore it is possible that trampling associated with activities other than periwinkle harvesting is causing the lower richness and diversity observed at Holy Island.

Natural and additional human-induced variation between rocky shores has the potential to mask impacts of intertidal harvesting, and when differences are detected it is challenging to separate the observed impacts caused by harvesting from all other co-existing coastal activities. To confidently detect changes from anthropogenic impacts, rocky shore communities need to be recorded multiple times a year to account for seasonal differences, and also over decades to see longer term trends (Hartnoll and Hawkins, 1980). Even then the risk remains that the provenance of major changes may be confused (Hartnoll and Hawkins, 1980).

5.4.4 Experimental study consideration

Ultimately, the experimental method of cage manipulations of periwinkles was not suitable within this study. The costs were too high, and the lack of relevancy for management if periwinkle stocks are not currently reduced was important.

Additionally, the effects of grazer exclusion have already been well studied on rocky shores (e.g. Petraitis, 1989; Cervin and Aberg, 1997; Lindberg *et al.*, 1998; Buschbaum, 2000; Cervin *et al.*, 2004) and other marine ecosystems (e.g. Hillebrand *et al.*, 2000; Lirman, 2001; Paine, 2002; Silliman and Bertness, 2002; Keuskamp, 2004; Poore *et al.*, 2012). Previous studies show a variety of impacts on rocky shore communities from reduced or extinguished grazer populations, such as: increased macroalgal germling survival (Cervin and Aberg, 1997), enhanced growth of grazer competitors (Petraitis, 1989), increased recruitment of sessile organisms (Buschbaum, 2000), and increased algal cover (AFBI, 2013).

These impacts are all possible for the Northumberland rocky shore communities if periwinkles were over-exploited to a level where their abundances were dramatically reduced. However, within Northumberland, there are at least two other common dominant grazers present on most shores: top shells (*Gibbula* spp.) and limpets (*Patella vulgata*), which may be capable of buffering the effects of reduced grazing pressure from *Littorina littorea*. This is a form of functional redundancy, whereby a species with an overlapping functional niche and distribution can be a substitute for the reduced species, ultimately maintaining ecosystem functioning and processes (Lawton and Brown, 1994; Rosenfeld, 2002). It is possible that in the long-term, reduced periwinkle density would have little impact on the community due to grazing competitors increased contribution. Several studies have observed dramatic alterations after short-term grazer exclusion, which have disappeared or changed in the long-term once other species alterations compensate (e.g. Lindberg *et al.*, 1998; Buschbaum, 2000). Therefore, lengthy exclusion studies are required to see realistic lasting effects over multiple seasons and years, which the scope of this thesis would not allow.

A study of direct effects could have also been considered, e.g. trampling or boulder turning. However, previously studies have found no changes using the simulated harvesting method for periwinkles, with background long-term harvesting levels

having the largest impact on communities (Crossthwaite, 2012). Therefore, it was decided that experimental methods were not required nor suitable within this study for periwinkle harvesting impacts. If in the future, evidence showed that periwinkle stocks were effected by harvesting, then manipulative field experiments may be more appropriate to infer community wide implications.

5.5 Conclusions

This chapter presents evidence which provides a baseline to help inform management plans for the BNNC EMS, as well as other protected areas throughout the UK. The evidence base provided is based on local, current harvesting levels, revealing actual impact (or lack thereof) of periwinkle collection activities within the BNNC EMS.

Periwinkles are important grazers and prey, with the ability to shape intertidal communities (e.g. Lubchenco and Gaines, 1981; Lubchenco, 1983; Watson and Norton, 1985; Petraitis, 1987; Janke, 1990; Vadas, 1992; Mill and Mcquaid, 1995; Anderson and Underwood, 1997; Sommer, 1999b; Buschbaum, 2000; Scheibling *et al.*, 2008; Griffin *et al.*, 2010; Diaz *et al.*, 2012). Results from this research suggest that impacts on periwinkles are not discernible against natural variability, and that at current, local collection intensities, periwinkle harvesting appears not to alter the periwinkle density or size beyond naturally occurring levels on sites with no collection pressure. However, it is possible that cumulative impacts over long timescales, or increased harvesting intensity in the future, could lead to negative impacts on periwinkle stocks in the future.

Similarly, it appears that at current levels, periwinkle collection is not causing observable negative impacts upon the rocky shore communities. The communities appear to be variable due to habitat differences and possibly alternative anthropogenic pressures, such as trampling (e.g. Povey and Keough, 1991; Brosnan and Crumrine, 1994; Fowler, 1999; Berthelon *et al.*, 2004; Tyler-Walters and Arnold, 2008; Ferreira and Rosso, 2009; JNCC and Natural England, 2011), rather than the occurrence or intensity of periwinkle harvesting.

Neither the target species nor rocky shore communities are clearly suffering in areas befalling harvesting. Similar to Boyes *et al.* (2006) review of threats from unlicensed marine activities, including hand gathering, there is insufficient evidence of detrimental impacts, despite the clear dangers.

Chapter 6 : Synthesis, Discussion, Management Implications, and Recommendations

Chapter 6. Synthesis, Discussion, and Management Implications

6.1 Synthesis

Management of fisheries, with the aim of ensuring sustainable exploitation, requires knowledge and understanding of the fishery itself (e.g. distribution of fishing pressure, exploitation level, etc.), and the potential and actual impacts incurred. A literature review of intertidal fisheries (Chapter 1), especially related to *Littorina littorea* and *Arenicola* sps, highlighted the importance of studying intertidal fisheries worldwide, and revealed many specific questions that remain concerning the scale, locale, and ecological impacts of these fisheries, both locally within Northumberland and nationally. This thesis aimed to explore details of the Northumberland periwinkle and lugworm fisheries. It has examined the scale, intensity, spatial distribution, drivers of fisher distribution, economic value, and associated impacts of both fisheries, using the BNNC EMS as a case study area. Social science and natural science methodologies have been combined to produce an integrated approach to the investigation, informing the current state of knowledge, and providing the first large scale fishery assessments for periwinkles and lugworms within England.

This chapter reviews the thesis, summarising the key findings in context of the literature, discussing the implications and potential uses for both local and wider management, recommending management actions, and finally highlighting outstanding research questions and priorities for future investigation.

6.2 Key Findings and Knowledge Contributions

6.2.1 Scale, Locale, Intensity, and Value

Growing concern for the exploitation of marine resources is leading to an increased demand for data on the distribution and intensity of fishing activities, including intertidal collection. Despite requirements for rigorous data to inform management (Dowling *et al.*, 2015a; Dowling *et al.*, 2015b), many fisheries remain data poor (Costello *et al.*, 2012), and to date, little research has been carried out for intertidal fisheries within England. Intertidal fisheries in general have received little attention compared to larger marine fisheries occurring in offshore and inshore environments (e.g. Phillips *et al.*, 2000; Arcos *et al.*, 2001; Drinkwater *et al.*, 2006; Stelzenmüller *et al.*, 2008; Abbott *et al.*, 2010; Bearzi *et al.*, 2010; Williams and Terawasi, 2011), including locally within Northumberland, where crab and lobster inshore fisheries have received significant recent attention (Turner *et al.*, 2009; Turner, 2010; Skerritt, 2014; Turner *et al.*, 2015; Stephenson, 2016). With very little data available on the spatial extent of the lugworm and periwinkle fisheries within Northumberland specifically, methodologies were developed to gather new data to assess the scale, locale, collection intensity, and economic value of both fisheries within the BNNC EMS (Chapters 2 and 3). The novelty of these investigations lies in the methods used, the local scale within a Marine Protected Area (MPA) setting, and the direct applicability to future management plans requiring evidence. Lugworm collection scale and intensity has been little studied within the UK, with only individual shores considered (Blake, 1979a), whilst periwinkle fishery studies have focussed on Scotland and Ireland in their entireties, resulting in less detailed analysis of larger areas (McKay *et al.*, 1997; Cummins *et al.*, 2002). This study bridged the gap of scales, combining the detail gained through studying individual shores with the 'bigger picture' approach of an entire coastline and EMS, providing large-scale, locally relevant data direct to conservation and fishery managers.

Chapter 2 combined spatial (shore observations/mapping) and social (fisher questionnaires) methods to explore patterns of lugworm collection on sediment shores, and periwinkle collection on rocky shores. Biomass removal was estimated and fisher distribution mapped for both target taxa. Clear collection hotspots were identified, with Boulmer being a key harvesting locale for both taxa, and a southern

skew of collection intensity within the BNNC EMS boundaries likely due to larger population centres occurring below the southern boundary (Northumberland County Northumberland County Council, 2014b). Seasonal patterns were revealed, with a winter peak in lugworm collection in line with the cod fishing season (Townshend and O'Connor, 1993; Fowler, 1999), and a summer peak for periwinkles due to a high demand for export (Cummins *et al.*, 2002). Biomass levels were significant for both fisheries, with a conservative estimate of 1.24 tonnes of lugworms, and 13.37 tonnes of periwinkles removed from the BNNC EMS each year, equating to economic values of £54,560 and £133,749 respectively. Lugworm biomass removal per standardised area of habitat was similar to other major bait fisheries around the world (ragworm at Dell Quay, Solent, and *G. dibranchiata* in Maine, USA) when the most popular shore (Boulmer) only was considered (Watson *et al.*, 2017a), providing evidence that the Northumberland lugworm fishery can be considered intensive in certain locations. The Northumberland periwinkle fishery appears to be less intensive than the Scottish or Irish counterparts (McKay *et al.*, 1997; Cummins *et al.*, 2002). The methods used and insights gained within this chapter have high applicability to other intertidal fisheries sharing the same habitats, with lugworms and periwinkles providing a case study for other species such as crabs, mussels, land lobster on the rocky shore, and ragworm, clams, and cockles on the sediment shores.

Adherence to existing lugworm management byelaws was found to vary considerably between location and designation reasons (i.e. practical vs conservation drivers), as has been seen before (Watson *et al.*, 2015), and suggesting that previously identified concerns about compliance and enforcement locally (NCAONB, 2009) are justified. Collection effort was further allocated into recreational and suspected commercial categories using key trends recorded in collection behaviour. Commercial collection appears to make up an overwhelming majority of both fisheries in terms of biomass, with 71% of lugworms and 95% periwinkles removed from the shores by suspected commercial fishers, providing further evidence that commercial intertidal collection must be assessed for management (Fowler, 1999; Watson *et al.*, 2017a), and providing a methodology to crudely separate recreational and commercial collection activities in a quantitative way.

Chapter 3 used spatial modelling methods to predict and describe the spatial patterns of lugworm fishing pressure, and the habitat and species sensitivities and vulnerabilities to bait digging. Two separate models were produced for lugworm collection suitability and sensitivity, which were combined to provide measures of vulnerability. This thesis is the first to use suitability modelling as a base for intertidal fisheries mapping and assessment. Data gathered to populate the model with lugworm density and size information is the first large scale assessment of lugworm populations within the UK, providing valuable lugworm population maps for the BNNC EMS. Output maps of lugworm collection suitability translated well into actual collection pressure (validated with shore observations from Chapter 2), providing a map of the Northumberland lugworm fishing grounds, an important aspect of Marine Spatial Planning (MSP) (Stelzenmüller *et al.*, 2008; Jennings and Lee, 2012), and increasing the understanding of the fishers and their choices, another important aspect of fishery assessment (Turner, 2010). Sensitivity maps highlighted key areas which would be most impacted if collection were to occur there. The vulnerability model relates the fishery pressure (suitability) to species and habitat sensitivity, highlighting key areas of conflict between the fishery and conservation aims. The areas with the biggest conflicts, and therefore likely the largest impacts, were identified as Fenham Flats, Budle Bay, Newton, and Boulmer. Current lugworm management (no-digging zones) spatially encompasses most of the areas identified as most suitable, sensitive, and vulnerable (UK Marine SACs Project, 2001a; NCAONB, 2009), with some spatial expansion recommended for improved coverage of vulnerable areas. This positive finding is dulled by the issue of enforcement; if areas are protected on paper only, they will have little helpful conservation effect.

6.2.2 Ecological Impacts

The main driver behind this thesis was to investigate the ecological impacts of the Northumberland lugworm and periwinkle fisheries. The policy shift behind this was DEFRAs 'Revised Approach to the Management of Commercial Fisheries in European Marine Sites' (MMO, 2014b). This insisted on an enhanced evidence base of the impacts of all fishery-interest feature interactions within protected sites, such as the BNNC EMS. Evidence of the impacts associated with intertidal fisheries are required for management decisions, as activities which unfavourably affect site integrity are not allowed without suitable management measures in place to protect

the interest features (MMO, 2014b), and impacts need to be known if they are to be minimised by conservation and management. Intertidal hand gathering activities on both rocky reefs and sand and mud flats were among the interactions identified as lacking an evidence base within the BNNC EMS (MMO, 2014b). Chapters 4 and 5 of this thesis addressed this evidence gap, using both comparative and experimental methodologies to explore the ecological impacts associated with the Northumberland lugworm and periwinkle fisheries, inferring whether these fisheries are compatible with the conservation objectives or the designated features of the BNNC EMS. The novel achievement of these investigations lies in their local, site-specific nature relative to the local fishing pressure and frequency, and the direct applicability of the findings to local marine management plans.

Chapter 4 investigated the impacts of the local lugworm fishery. Previous studies have revealed that possible impacts of bait collection include altered density and size structure of the target species populations (Shahid, 1982; Beukema, 1995; Volkenborn and Reise, 2007), and reduced biomass and altered community structures of associated infaunal organisms (Jackson and James, 1979; Van den Heiligenberg, 1987; Brown and Wilson, 1997). Methods were developed to detect similar changes within the BNNC EMS, where no historical data was available to observe the changes over time with the presence of harvesting. The target species populations and the associated sediment communities were compared both between sites experiencing contrasting lugworm harvesting pressures (actual fishing pressure), and within a single site before and after experimental bait digging disturbance events and lugworm exclusion/reduction (simulated fishing pressure).

The comparisons between sites revealed that the target species populations were relatively similar, and there was no evidence that populations were suffering at sites befalling heavy collection. This does not mean that changes have not occurred over time which are undetectable against natural variation between sites. In contrast, there was strong evidence that the sediment community was impacted by the occurrence of lugworm collection, especially at Boulmer, the high fishing pressure site, where average infaunal abundance was less than half that of the other sites. Despite the observed decreased infaunal community with increased collection pressure, the diversity was not negatively impacted.

Simulated digging within a previously undisturbed site revealed that bait digging for lugworms within Northumberland sediment shores has the potential to dramatically impact the infaunal community, with reduced abundance, taxonomic richness, and altered community structure, providing further evidence in support of the differences observed between sites being due to the lugworm fishing pressure. The higher the digging frequency/intensity the larger the impacts observed, which has been suggested previously (Van den Heiligenberg, 1987; Beukema, 1995). Alterations to the habitat were also recorded, with the sediment penetrability severely increased after digging. Reducing lugworm abundance marginally with exclusion nets was seen to impact indirectly on the sediment community, reducing the taxonomic richness. Previous lugworm exclusion studies have excluded all lugworms from an area (e.g., Volkenborn and Reise, 2006; Volkenborn *et al.*, 2007a; Volkenborn and Reise, 2007; O'Brien *et al.*, 2009; Kuhnert *et al.*, 2010a; Lei *et al.*, 2010), and as such this is the first study to show that even a slightly reduced lugworm abundance can affect communities, due to the important ecological role of lugworms as ecosystem engineers (Lawton, 1994; Wright and Jones, 2006; Volkenborn *et al.*, 2007a; Volkenborn and Reise, 2007).

Experimental plots recovered rapidly, with full infaunal recovery occurring within 11 weeks (previous studies range from 140 days to 5 years (Van den Heiligenberg, 1987; Beukema, 1995). It is unlikely that recovery would be as complete and as fast at sites experiencing significant fishing pressure. Unique conditions at Fenham Flats allow optimal recovery, for example, large expanse of proximate undisturbed sediment for adult migration, small scale of disturbance, few long-lived and slow-recovery species present, recovery period in summer when recruitment peaks, etc. (Jackson and James, 1979; Zajac and Whitlatch, 1982a; Zajac and Whitlatch, 1982b; Beukema, 1995; Ford *et al.*, 1999; Fowler, 1999; Reise *et al.*, 2001; Watson *et al.*, 2007)). Although a positive sign, the fast recovery rate observed in this study should not be applied to all other sites within the BNNC EMS for which recovery could be significantly slower depending on the local conditions. Marginal sites such as the bait digging zone within Fenham Flats can be assumed to have similar recovery potential if digging effort is spread out, however, other sites must be considered on an individual basis.

Chapter 5 explored the impacts associated with the Northumberland periwinkle fishery. Previously identified impacts of rocky shore gastropod harvesting are similar to those of bait digging, with possible impacts including a reduced stock and body size of the target species, and altered community interactions and structure (Petraitis, 1989; Cervin and Aberg, 1997; Quigley, 1999; Buschbaum, 2000; Thompson *et al.*, 2002; Berthelon *et al.*, 2004; Crossthwaite, 2012). A comparative methodology was used to identify any observable impacts of these kinds between rocky shores within the BNNC EMS which are subject to a gradient of periwinkle harvesting intensities. There was no evidence that the target species is negatively impacted at current harvesting intensity, with higher average abundances observed at collected sites, and body size patterns not corresponding to collection pressure. The rocky shore community showed a similar lack of impacts, with Holy Island, the low intensity collection site, having the most distinct community and lowest species richness and diversity. Previous research has suggested that impacts upon rocky shore target species and communities are cumulative, requiring long timescales to see the effects, with historic collection patterns capable of masking the present impacts (Crossthwaite, 2012). Therefore, although this study did not observe any impacts, they could be masked by natural variation between sites, unknown historic collection patterns, and other disturbance events. Unfortunately no historic data is available to explore changes over time.

Overall, it appears that at current harvesting intensities the Northumberland lugworm fishery is not damaging target species populations, but having significant negative impacts upon the infaunal community at high harvesting intensities. The Northumberland periwinkle fishery in creating no observable negative impacts on either the target species or the rocky shore communities. Impacts of this nature are often cumulative and may begin to appear with continued or increased collection in the future, so ongoing monitoring is recommended for both fisheries.

6.3 Implications for management

6.3.1 BNNC EMS

Despite the requirement for mudflats, sandflats, and rocky reefs within SACs to be maintained in a favourable condition (European Union Council Directive. 92/43EEC, 1992), intertidal fisheries management remains an issue for regulatory bodies. Management concerns over the severe lack of impact evidence of intertidal fisheries was the driver of this research (MMO, 2014b), and as a result the findings are aimed to be directly applicable to NIFCA and Natural England marine management plans, increasing the knowledge base on lugworm and periwinkle intertidal fisheries within the BNNC EMS. Current trends towards the decentralisation of fisheries monitoring and management, and increasing regional responsibility (Gavaris, 1996; McCay and Jentoft, 1996; Jentoft *et al.*, 1998; Lewins *et al.*, 2014; Eliasen *et al.*, 2015) require regionally specific data, which this thesis provides.

This research has successfully developed methodologies to map the distribution of intertidal fishing activities (either modelling in Chapter 3, or shore observations in Chapter 2), and to estimate the fishing intensity in terms of biomass and economic value (Chapter 2), which were previously lacking within Northumberland. These methods can be used and repeated by managers responsible for assessing other local intertidal fisheries, and will continue to be used by NIFCA and Natural England to monitor periwinkle and lugworm fisheries distribution and scale over time.

Collection observations in Chapter 2 suggest that both lugworm and periwinkle collection are spatially patchy, which could be considered 'self-limiting', creating natural no-take zones. The lack of collection observed in some areas could create refuges for unexploited populations, which if large enough, could help to sustain exploited populations (Carr and Reed, 1993). Both target species larval stages have high dispersive potential (Johannesson, 1988; Günther, 1992; Jackson, 2008a; Tyler-Walters, 2008), making it possible for refuge populations to supply new recruits to harvested areas (Carr and Reed, 1993). This is a positive for managers if refuge populations exist naturally without the need for further management. However, it is essential to determine the effectiveness of such refuges, ensuring that they have the ability to maintain high larval production and replenish stocks at a level higher than the established harvesting rate (Carr and Reed, 1993).

Compliance of fishers to bait digging byelaws was observed to be low in some areas, such as Newton, where night time collection was high in order for fishers to evade enforcement (and Regulations were scored to be of low importance by experts within the suitability model weightings). The prevalence of night time collection and its associated darkness poses a major difficulty for management, creating additional practical challenges compared to day time collection (Cooke *et al.*, 2016). It is important for resource management agencies to decide if and how they manage night fisheries (Cooke *et al.*, 2016). A major management implication of these fisheries is the requirement to observe and enforce at night, when logistics can be challenging – safety, staffing effort, etc. (Cooke *et al.*, 2016). Effective enforcement is critical to achieve a high level of compliance (Ceccherelli *et al.*, 2011; Cooke *et al.*, 2013; Watson *et al.*, 2015), an important issue in marine management (e.g. Burger *et al.*, 1999; Gezelius, 2002; Crawford *et al.*, 2004; Hatcher and Gordon, 2005; Blank and Gavin, 2009; Bloomfield *et al.*, 2012; Haggarty *et al.*, 2016). Strong, consistent, face-to-face enforcement is imperative, with passive methods such as education, codes of conduct, and signage largely ineffective (Watson *et al.*, 2015). For the Northumberland bait worm fishery, this would require significant additional resources to enable enforcement a night. However, this studies collection pattern findings could help to direct limited enforcement resources to maximise effectiveness – collection hotspots, seasons, and tidal states. To increase compliance of current spatial management, it is recommended that enforcement is increased, particularly at night when a high amount of illegal fishing activity often occurs (e.g. Anderson, 1989; Crawford *et al.*, 2004; Ganapathiraju, 2012). The observed southern skew of collection hotspots could aid enforcement bodies such as NIFCA, based in South-East Northumberland, reducing travel time and costs.

The distinction of commercial from recreational intertidal fishers is a challenge (Griffin, 1988; Fowler, 1999), and remains a topic of debate (Watson *et al.*, 2017a). In this study, commercial collection appears to dominate both fisheries in terms of biomass. For management to focus solely on commercial collection, which is often the case for bait fisheries (Watson *et al.*, 2017a), the identification of commercial collectors needs to be achievable on-site. This remains difficult, unless commercial collection is given an agreed definition based on observable characteristics, such as harvest amount (recommended at over 200 lugworms and 20 lbs periwinkles per

collector per trip from this study). Watson *et al.* (2017a) argues that management targeting commercial collection only will fail due to the uncertainties with identifying commercials, resulting in ineffective management. A possible solution would be to invoke management for both sectors, focussed on reducing the high intensity collection most commonly associated with commercial collection.

Monitoring is the action of intermittently recording the condition of a feature to measure or detect compliance with a predetermined standard (Hellawell, 1991). It is an essential element of management (Day, 2008), which is used to: inform conservationists when the system departs from the desired state, detect the effects of impacts and disturbances, and measure the success of management actions (Legg and Nagy, 2006). Baseline data is required to measure against to detect change (Goldsmith, 2012). This study provides the first BNNC EMS wide dataset of lugworm populations, which can now be used as a baseline to measure future stock changes, with either continued collection impacts, or altered management (Stelzenmüller *et al.*, 2008). Lugworm density is generally stable over time (long-term) in unexploited stocks compared to other infaunal organisms (Beukema, 1982), which should make detecting human induced changes easier. There is seasonal variation, with the highest densities in spring/summer (Brey, 1991), and therefore monitoring should take place at the same time each year to avoid natural variation between seasons influencing the data-set (Hewitt and Thrush, 2007). Ongoing monitoring should fall within April/May to coincide with the baseline data provided in this study, whilst also avoiding possible interference during the peak bait digging season in winter (Fowler, 1999).

Monitoring and research design are often a compromise between the scientific ideal, and financial and logical constraints (Warwick, 1993; Gerber *et al.*, 2005). Good value monitoring methods are important to consider in management plans, reaching a reasonable balance between cost and the quality of knowledge gained, leading to better management (Gerber *et al.*, 2005). One such compromise for monitoring lugworm populations could be to reduce the spatial extent of monitoring, whilst maximising the ability to detect changes (Manley *et al.*, 2004; Nichols and Williams, 2006). Monitoring could be directed spatially using the model output maps produced in this study, for example to areas where collection activities are likely to have the largest impacts (high vulnerability), or areas where collection pressure is highest

(high suitability), since bait digging impacts are known to increase with collection intensity (Anon, 1992 as cited by JNCC and Natural England, 2011). Additionally, areas of sediment with low suitability (likely largely uncollected) could be used as controls, observing the natural variation in lugworms and the associated species and habitats (Block *et al.*, 2001).

Current spatial management of bait digging covers the majority of the most suitable, sensitive, and vulnerable sites to lugworm collection identified by the models in Chapter 3, suggesting that existing management within the BNNC EMS is well placed to protect the interest features from potential lugworm harvesting impacts. Berwick north of the pier, a very suitable shore, and Boulmer north of the no digging zone, a very vulnerable area are the two exceptions, and would both benefit from additional management measures, such as spatial closure, to further improve the management coverage of areas highlighted within the models. Fisheries displacement if all collection was disallowed and well enforced at Boulmer and Berwick would likely effect shores with the highest suitability close by (Underwood, 1993; Boye *et al.*, 2006; Abbott and Haynie, 2012; Lédée *et al.*, 2012), such as parts of Foxton, Howick, Newton, Eyemouth, and Fenham Flats bait digging zone. Failure to consider fishers behaviour, such as relocation, within management policies is an error, and can undermine the success of the measures implemented (Rosenberg and Restrepo, 1994; Wilen *et al.*, 2002). Models can be used to predict fisher relocation patterns and costs (Dowling *et al.*, 2012). The suitability model from this study has potential to be used as a dynamic management tool, by testing spatial management scenarios for collection displacement, providing a method for 'management strategy evaluation' (Smith *et al.*, 2007). Potential spatial closure areas could be removed from the model, leading to an altered spread of suitability scores across the BNNC EMS.

The impact study findings (Chapter 4 & 5) reveal to managers that the Northumberland lugworm fishery is currently having a negative impact upon the sediment community only. Infaunal communities are an important sub-feature of the BNNC EMS (European Union Council Directive. 92/43EEC, 1992), and as such should be protected from damage. Local management to reduce the impacts on the infaunal communities may be required. No impacts were recorded from the periwinkle fishery, suggesting that current collection levels are not damaging to the

rocky reef interest features of the BNNC EMS, and as such are unlikely to require further management measures, although ongoing monitoring is recommended.

6.3.2 Wider Implications

Although regional specificity was the intention and a novelty of this thesis, the findings are nevertheless pertinent to the wider management of intertidal fisheries. Some of the insights gained and the methods used in this study have wider relevance and some generalisation is possible; the BNNC EMS acts as a case study for lugworm and periwinkle fisheries nationally, and intertidal fisheries in an international context.

Marine managers strive to ensure sustainable resource use, and evade conflicts over space and limited resources (Jennings and Lee, 2012). Marine Spatial Planning (MSP), an emerging place-based method of ecosystem based management (Crowder and Norse, 2008), often utilises Geographical Information Systems (GIS) to study complex interactions in marine and coastal areas (Douvere, 2008). If intertidal fisheries are to be incorporated into MSP, they need to be spatially represented in a suitable format. This study has successfully developed and tested combined approaches of shore observations and suitability modelling to spatially define intertidal fishing grounds, the outputs of which are suitable for incorporation into MSP projects.

There has been an emphasis on ecosystem approaches to management in recent years (Douvere and Ehler, 2007), whereby multiple pressures are considered together. However, individual activities and impacts require assessment before data can be combined into ecosystem wide outlooks (Purcell *et al.*, 2010). This study explores and models lugworm collection, within a discreet area. However, the findings from this study have the potential to be incorporated into larger MSP projects (Shucksmith *et al.*, 2014), alongside other fisheries (e.g. *Nereis* spp) or shore use and activity data, providing a more complete spatial picture of intertidal exploitation in the light of shore use complexity. For example, the MMO recently produced a marine recreation model, incorporating maps of multiple activities to produce an overall picture of recreational marine use (MMO, 2012). The modelling methods used within Chapter 3 can also be adapted and applied to multiple other

intertidal target species, as well as lugworms in other localities. Lugworms were an ideal test organism due to the ease and non-intrusive nature of counting and measuring casts on the sediment surface (Flach and Beukema, 1994) compared to sediment sampling and sorting (e.g. for ragworms (Watson *et al.*, 2007)). Multiple detailed individual species fishery distribution models could be combined to inform a multispecies management approach (May *et al.*, 1979).

The spatial methods used in this thesis can be easily and effectively adapted to include alternate target species, and repeated in various geographic locations. The suitability modelling method used in Chapter 3 is a cost effective way to map fishing grounds for fisheries which are data poor, especially for intertidal fisheries where no formal recording methods are practiced (e.g. logbooks and vessel tracking systems, which are common place in inshore and offshore fisheries (Jennings and Lee, 2012)).

Methods developed in this study to assess fisheries impacts in areas lacking baseline data also have great applicability to other fisheries and localities. The methods used to compare collected and uncollected areas are similar to those used in numerous studies in managed marine reserves or no-take zones versus unprotected areas (e.g. Keough and King, 1991; Keough *et al.*, 1993a; Shears *et al.*, 2006; Lester *et al.*, 2009). However, here they are adapted for a region where less clear distinctions can be made, for example no binary designation of 'harvested' or 'not harvested', leading to the assignment of a collection pressure gradient. This adaptation of a common comparative methodology (FAO, 2005) allows assessments to be made in areas lacking clearly defined and adhered to protected areas, and scarce baseline data, which is often the case for intertidal fisheries. Comparative methods are imperative to study 'actual' impacts from realistic harvesting intensities and scales (FAO, 2005) where data over time is not available, and are valuable not just within highly protected marine reserves.

This thesis has further demonstrated that spatial and social aspects of fisheries are complex and two intertidal fisheries within the same area can have very different characteristics. For example, the Northumberland periwinkle fishery is well spread throughout the study area, whereas the Northumberland lugworm fishery is more spatially focussed, and the seasonality contrasting. The need to increase

understanding of the cumulative and interactive effects of multiple environmental stressors, such as fisheries, is well documented and acknowledged (Crain *et al.*, 2008), however the large contrasts between these two intertidal fisheries highlights the requirement to study all fisheries individually and fully before incorporating intertidal fishing more generally into management plans and MSP. The data requirements for ecosystem-based fisheries management are large, requiring individual fishery data for multiple species before interactions are considered (Latour *et al.*, 2003). Fisheries around the world must be studied in more detail to fully understand the patterns, drivers, and impacts, if they are to be managed successfully.

One general lesson derives from the issues observed with adherence to byelaws, and the associated requirement for effective enforcement. Compliance has been studied for many fisheries (e.g. Burger *et al.*, 1999; Gezelius, 2002; Crawford *et al.*, 2004; Hatcher and Gordon, 2005; Blank and Gavin, 2009; Bloomfield *et al.*, 2012; Haggarty *et al.*, 2016), and effective enforcement is already widely acknowledged as imperative for successful marine management. This study adds to the evidence base that increased enforcement effort may be key to tackle issues of non-compliance, with additional face to face contact a possible improvement (Watson *et al.*, 2015). It particularly highlights the requirement of night time enforcement for intertidal worm fisheries, if designated areas are to be fully protected from bait digging.

The background research of this thesis presented in Chapter 1 exposes and highlights the lack of control and regulations in intertidal fisheries generally. The difficulty in evaluating intertidal fisheries such as those covered in this study, is the severe lack of data currently available. Landings data is not required by law (Fowler, 1999; Cummins *et al.*, 2002) and when it is available, it is often unreliable due to the unregulated nature (McKay *et al.*, 1997; Cummins *et al.*, 2002). Many inshore and offshore fisheries are subject to stricter regulation, such as logbooks and landings declarations (MMO, 2014a). This thesis has provided evidence that local Northumberland harvests are significant for two intertidal species, suggesting that fisheries data recording is important for intertidal fisheries in general, not just inshore and offshore variants. Intertidal fisheries would benefit from more regulated recording, with commercial harvest quantities and locations legally required.

6.4 Management Options, Recommendations, and Responsibility

Both periwinkle and lugworm fisheries are currently minimally managed and largely unassessed when compared to other, often higher value fisheries (e.g. McKay *et al.*, 1997; Fowler, 1999; Cummins *et al.*, 2002; Watson *et al.*, 2017a). The public right to collect bait for personal use is often seen as a major obstacle in marine management, only fully diminishable by an Act of Parliament. However, public rights can be justifiably controlled by statutory bodies or competent authorities under a range of legislation (Bean and Appleby, 2014) to prevent ecological damage to designated features. Possible management methods include: voluntary guidelines and codes of conduct, byelaws for closed areas, several orders, regulating orders, licencing, weight or bag limits, size limits, and closed seasons (Underwood, 1993; UK Marine SACs Project, 2001c; Harthill *et al.*, 2005; Boye *et al.*, 2006; DEFRA, 2012; AFBI, 2013). All methods have advantages and disadvantages (discussed in Chapter 1), and it is suggested that these are considered on a case by case basis, looking at the relevant scientific evidence to fully inform management decisions (Bean and Appleby, 2014).

6.4.1 Management Recommendations – Improving Existing Management

Currently within the BNNC EMS, the periwinkle fishery is completely unmanaged, whilst the lugworm fishery is controlled with 'no digging byelaws' located at Holy Island, Newton, and Boulmer.

Overall, the current spatial management appears well placed to protect the most vulnerable habitats and communities from lugworm collection. Modelling lugworm collection in Chapter 3 highlighted just two key shores which were either highly suitable or highly vulnerable, which are not currently encompassed by regulations. Berwick (north of the pier) and Boulmer (south half of the shore) are areas which could benefit from the expansion of current spatial regulations. It is recommended that Berwick be considered for no take regulations in the future. A zonation pattern such as that seen at Boulmer (where only half the shore is a no digging zone) may be the best option, allowing collection to continue, whilst also protecting the local habitat, infaunal community, and lugworm population from overexploitation.

Enforcement of current lugworm management is split between three separate bodies. Natural England Lindisfarne NNR manager is responsible for enforcing at

Holy Island, National Trust wardens enforce the no digging byelaw at Newton, and NIFCA observe the no digging area at Boulmer. Non-compliance with these regulations is a serious issue identified within this study (especially at Newton). Improved enforcement methods generally (more regular face-to-face presence and contact), and the addition of night time observations is recommended if conservation managers want to ensure and enhance the effects of current management methods. To make the most of limited resources, enforcement effort should be directed towards spring low tides in January and February, both day and night where possible. To enable more thorough enforcement with limited resources, remote electronic monitoring techniques such as closed circuit television or drones could be considered (Mangi *et al.*, 2015; Wright, 2015; Lord, 2017), enabling managers to view collection activities on a larger, less targeted scale (both spatially and temporally).

6.4.2 Management Recommendations – Exploring New Management Options

The main impacts observed in this thesis were on the infaunal sediment communities and the habitat, rather than the target species (lugworms and periwinkles), and as a result the recommendations made here reflect this.

Table 6:1 shows the various management options with reasoning and rationale as to what is recommended or not recommended for each fishery, in light of the results of this study. For the lugworm fishery, the highest recommended management method is closed areas, both continued and new additions. Additionally, closed seasons to enhance community recovery, and bag limits to control commercial collection are considered appropriate, as is a bait digging code of conduct as long as education and enforcement are prioritised. For the periwinkle fishery, only size limits are recommended as a precautionary approach to management.

The management discussed could help to reduce the observed impacts and limit future damage, however, management methods need to be scientifically tested for effectiveness before they are fully recommended or implemented, taking into account both social and economic issues as well as ecological benefits (Watson *et al.*, 2007).

Table 6:1: Management Recommendations for both the Northumberland lugworm and periwinkle fisheries with reasoning based on the findings of this thesis. Recommendation scores range from 0-10, with 0 being 'not recommended' and 10 being 'strongly recommended'.

Management Method	Lugworm Fishery Recommendation Score?	Periwinkle Fishery Recommendation Score?	Reasoning
Voluntary Guidelines and Codes of Conduct	6	0	· Lugworm - Codes of conduct ensuring back-filling practices could reduce the severity of infaunal mortalities if followed (Fowler, 1999). However, code of conduct compliance issues have been observed in other fisheries (Watson, 2014; Watson *et al.*, 2015), and as such this avenue is only worth pursuing if education and enforcement is active. · Periwinkle – No impacts observed to have codes for. If in the future impacts on the rocky shore communities were observed, codes of conduct ensuring the repositioning of boulders after turning could be advantageous (Davenport and Davenport, 2006).
Byelaws / Closed Areas	10	0	· Lugworm – 3 areas already covered by byelaws, with more recommended to extend coverage to include Berwick. Areas which are less suitable but highly sensitive to lugworm collection activities (model outputs from Chapter 3) could also be closed to protect more 'pristine' sediment communities from occasional or opportunistic digging (with little backlash from collectors and anglers since they are not popular shores), resulting in pockets of natural infaunal communities available to restock impacted shores (Di Lorenzo *et al.*, 2016). Shores could be chosen for closure if they are especially diverse and important examples of high biodiversity. · Periwinkle – No impacts observed to close areas for.
Licensing / Permits	3	0	· Lugworm – The main advantage of this method is the monetary gain which would allow for better enforcement of current and new management regulations, and the ability to attach further conditions, such as limit the number of collection days, etc. (Boye *et al.*, 2006). However, this would require high policing to ensure permit use was adhered to. May be unnecessary when no target species impacts have been observed. · Periwinkle – No impacts observed to require permitting
Weight or Bag Limits	7	0	· Lugworm – Although no lugworm abundance impacts were observed at current harvesting intensities, limiting the amount each individual is allowed to collect could help with limiting illegal commercial collection (stopping commercial collectors harvesting over 500 worms regularly). However, from a conservation perspective, it would not stop habitat destruction and

			disturbance for the infaunal community (Underwood, 1993), so would need to be combined with other conservation aimed management measures. · Periwinkle - No Impacts observed on target species abundance related to over harvesting
Size Limits	0	5	· Lugworm – No Impacts seen on lugworm size from collection, and don't stop physical disturbance. Also collectors already preferentially take the larger individuals (personal communication), and nursery beds are located separately so juveniles are not usually disturbed (Fowler, 1999). Would also be very difficult to police – would need measurements to be taken on site. · Periwinkle – Despite the lack of impacts observed on periwinkle size in this study, many other regions already have minimum landing sizes for local periwinkle fisheries, which is always 16mm shell height (Stranford Lough & Lecale Partnership, 2013). This would be easier to police for the periwinkle fishery, where most of the sales go through wholesalers who could be responsible for enforcing the regulations (no money for undersized individuals). This could act more as a preventative conservation measure within the BNNC EMS.
Closed Seasons	7	0	· Lugworm – Appropriately timed closed seasons could help to improve the recoverability of infaunal communities. Ceasing continued sediment disturbance during key reproductive periods (spring) could enhance recovery (AFBI, 2013). Closing popular shores to collection in spring would allow maximum community recovery to occur, whilst minimally impacting collectors compared to winter closures (less popular collection period). · Periwinkle – No impacts observed to require closure.

6.4.2 Management Responsibility

Responsibility for implementation and enforcement of marine management plans is often debated. Bean and Appleby (2014) reviewed the potential regulators and regulation options in relation to the relevant legislations in detail for the Welsh bait and seaweed fisheries, of which most is applicable to the BNNC EMS intertidal worm and shellfish fisheries. The most obvious regulators are: the Inshore Fisheries and Conservation Authorities (IFCA's) under the legislation of the Marine and Coastal Access Act 2009, Natural England under the legislation of the Wildlife and Countryside Act 1981, or the local County Council under the legislation of the Public Health Acts Amendment Act 1907 (Bean and Appleby, 2014). It is important that the responsible authorities are widely accepted and acknowledged to allow for appropriate management to be put in place in a timely fashion.

Northumberland Inshore Fisheries Conservation Authority (NIFCA), Natural England, and Northumberland County Council have all already implemented bait digging regulations within the BNNC EMS, in the form of byelaws (Natural England no digging zone at Budle Bay and Fenham Flats, NIFCA seagrass protection byelaw, and Northumberland County Council no digging zone at Boulmer) (UK Marine SACs Project, 2001a; NCAONB, 2009; NIFCA, 2013a), suggesting that all three regulators are appropriate options for further management measures if required. There are currently no regulations in place for periwinkle collection within Northumberland, however, in other areas of the UK, collection regulations for periwinkles (e.g. minimum landing sizes and closed seasons) have been set by multiple IFCA's (Stranford Lough & Lecale Partnership, 2013), suggesting that NIFCA may be best suited to regulate periwinkle collection within the BNNC EMS.

6.5 Limitations and Future Research

The lack of local baseline data for the target species stocks and the associated communities led to difficulties in assessing fisheries impacts over time. To overcome this, comparative methods of collected and uncollected sites were used, but this method came with additional limitations: the presence of natural spatial variation between sites, potentially masking observable impacts (Borcard *et al.*, 1992). Regional variation between intertidal communities and habitats (e.g. Fraschetti *et al.*, 2001; Ysebaert and Herman, 2002; Fraschetti *et al.*, 2005) also means that results are not directly applicable to other localities, even if collection pressures are very similar elsewhere. This study provides in depth and large scale baseline data, so that future studies within the BNNC EMS can monitor changes in the target species and the communities over time (Spellerberg, 2005; Goldsmith, 2012), without the caveat of spatial variation.

The extent of data collection, both temporally and spatially, is a further limitation of this thesis. Data collection were limited by time, finances, and logistics for several chapters, common limitations of short-term PhD research. Biomass removal, collection hotspots, and byelaw compliance estimates were limited temporally, with twelve months of data collection being adequate to observe seasonal but not inter-annual patterns and variation (Lynch, 2014), resulting in a 2014-2015 snap-shot of fishing effort. Inter-annual patterns of lugworm and periwinkle collection should be explored in future studies, to further improve the knowledge base of the fisheries. The spatial confinement of this study within the BNNC EMS administrative boundaries was critical to answer local management questions, however, understanding the wider fishing effort and impacts is important (Piet and Quirijns, 2009) and may help to interpret relationships. Repetition of some of the methods outside of the EMS is desirable, especially South Northumberland and Tyne and Wear where collection effort is high anecdotally, and could help to inform future management designations (Kelleher and Kenchington, 1991). A further spatial limitation occurred during the collection of lugworm distribution and size data for the spatial model, with less accessible areas missing, which could be added to the model at a future date to improve the accuracy of the output for those areas. The whole modelling process could also be repeated for the periwinkle fishery in the future.

An ecosystems based approach to marine management is becoming a widely accepted tactic, whereby numerous pressures are considered together (Douvere and Ehler, 2007; Douvere, 2008). This thesis examines the local lugworm and periwinkle fisheries only, overlooking all other intertidal fisheries, such as ragworms, shore crabs, etc. (Fowler, 1999), which overlap spatially within the BNNC EMS (personal observations). The findings of this study can be combined with those of other such fisheries as they become available, and the methods used within this thesis can be adapted for those target species. Incorporating data from multiple intertidal fisheries, accounting for shore-use complexity, is recommended for future research, as it is difficult to isolate and separate the impacts from fisheries with similar community wide impacts (e.g. rocky shore trampling (Berthelon *et al.*, 2004; Tyler-Walters and Arnold, 2008) from periwinkle and crab collectors).

Additional data for robust stock assessment and evaluations of sustainability are still lacking for all intertidal fisheries within the BNNC EMS. Stock identification (Begg *et al.*, 1999), stock size and spawning stock size (Hilborn and Walters, 2013), current and optimum fishing mortality rates (Walters and Martell, 2002), maximum sustainable yield (Mace, 2001), and stock-recruitment relationships (Lee *et al.*, 2012), could all be investigated for intertidal fisheries towards the creation of stock assessment models, with the aim of maximising fishery sustainability.

Data gaps remain for local lugworm and periwinkle fisheries, as well as other intertidal fisheries occurring within the BNNC EMS, as discussed, but this thesis has made a major contribution to the knowledge base of both fisheries. It demonstrates that intertidal fisheries should and can be studied effectively with limited resources, and provides methods to roll out fisheries assessments for other species and localities.

6.6 Summary

Previous Northumberland fishery research has focussed on the larger and more valuable inshore and offshore fisheries such as shellfish (e.g. Turner *et al.*, 2009; Turner, 2010; Skerritt, 2014; Turner *et al.*, 2015; Stephenson, 2016), with significant knowledge gaps within the intertidal fisheries sector. The Northumberland periwinkle and lugworm fisheries have never before been studied in detail at a regional scale, and data was severely lacking to inform management bodies. This thesis provides the first large scale assessment of two Northumberland intertidal fisheries, accounting for both spatial patterns relevant to MSP and impact assessments relevant to management questions. It offers fresh, regionally specific insights into the collection of periwinkles and lugworms within the BNNC EMS, with scope for future methods application for various target species and other regions or MPAs. Despite the scope for transferability of methods or insights nationally or even internationally, the main use of the data presented in this thesis should be to inform current local management questions raised by the recent 'Revised Approach to the Management of Commercial Fisheries in European Marine Sites' (MMO, 2014b), and aid future management by providing a comprehensive baseline and methods for monitoring, allowing changes to be assessed over time.

Results from this research suggest that the periwinkle and lugworm fisheries occurring within the BNNC EMS are having little impact on the target species density at current harvesting intensities, at the sites within the short window of the study. However, it has been demonstrated that lugworm collection appears to be damaging the habitat and associated infaunal communities to some degree (Chapter 4 & 5). Natural spatial variability in the target species populations and communities may be masking additional impacts, such as reduced target species size classes, and focussed monitoring over time is recommended to observe changes in populations with continued collection pressure. Although lugworm and periwinkle populations appear equally robust at all sample sites, with average densities greater than 13 lugworms and 4 periwinkles per meter square at all shores regardless of harvesting intensities, no conclusions are made with regard to the overall sustainability of the fisheries. The presentation of fishing grounds and biomass and economic value estimates (Chapters 2 & 3) provides marine managers with spatial data at a relevant local scale for incorporation into complex MSP, and highlights the importance of both

fisheries locally, and in turn the need for appropriate regionally specific resource management to sustain harvests long-term.

In light of the findings of this thesis, recommended management for the Northumberland lugworm fishery includes: increased patrolling and enforcement effort of current regulations (especially at night), monitoring of the target species and habitats over time (using baselines provided), maintenance of current closed areas with increased spatial coverage, closed season in spring to aid recovery at popular sites, bag limits to counter illegal commercial activity, and a bait digging code of conduct to minimise damage. Recommended management for the Northumberland periwinkle fishery includes: monitoring of the target species and habitats over time (using baselines provided), and size limits as a precautionary approach.

Overall, despite many areas of research remaining, this thesis has made a contribution to the study of intertidal fisheries, using a mixed methods approach to reach an interdisciplinary understanding. It is recommended that the approaches used are developed further, and with continued use, increase understanding of local intertidal fisheries beyond the scope of this thesis, enabling the development of fully informed regionally appropriate management plans.

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Appendix A: Questionnaires

Lugworm Collection Questionnaire

Date:

Location:

1. Which shores within North Northumberland do you collect lugworms from?
   - ☐ Boulmer
   - ☐ Newton
   - ☐ Alnmouth
   - ☐ Howick
   - ☐ Beadnell
   - ☐ Embleton
   - ☐ Seahouses
   - ☐ Bamburgh
   - ☐ Holy Island
   - ☐ Scremerston
   - ☐ Other:

2. Which other shores that you know of are collected in the area?
   - ☐ Boulmer
   - ☐ Newton
   - ☐ Alnmouth
   - ☐ Howick
   - ☐ Beadnell
   - ☐ Embleton
   - ☐ Seahouses
   - ☐ Bamburgh
   - ☐ Holy Island
   - ☐ Scremerston
   - ☐ Other:

3. How often do you collect?
   - ☐ Daily
   - ☐ Every other day
   - ☐ Twice a week
   - ☐ Weekly
   - ☐ Fortnightly
   - ☐ Monthly
   - ☐ Every few months

4. How long per tide do you collect for?
   - ☐ Less than 1 hour
   - ☐ 1-2 hours
   - ☐ 2-3 hours

☐ 3-4 hours
☐ 4-5 hours
☐ More than 5 hours

5. How many lugworms do you collect in this time?
☐ 0-50
☐ 50-100
☐ 100-150
☐ 150-200
☐ 200-250
☐ 250-300
☐ 300-350
☐ 350-400
☐ 400-450
☐ 450-500
☐ 500-550
☐ 550-600
☐ 600-650
☐ 650-700
☐ More than 700

6. What months do you collect in?
☐ January
☐ February
☐ March
☐ April
☐ May
☐ June
☐ July
☐ August
☐ September
☐ October
☐ November
☐ December

7. Which days do you collect?
☐ Weekends
☐ Bank holidays
☐ Weekdays

8. Does the low tide height effect when you collect (e.g. spring or neap tide)?
☐ Yes – I only collect at the lowest tide heights (e.g. spring tides)
☐ No – I collect at any height of low tide
☐ Yes/No – I try to collect at the lowest tide heights, but I also collect at higher low tide heights as needed

9. Do you collect in the dark?
   - ☐ Yes – But just early mornings and winter evenings
   - ☐ Yes – Including the middle of the night (after midnight and before 4am)
   - ☐ No – I only collect in daylight hours

10. What tool do you use to collect lugworms?
    - ☐ Fork
    - ☐ Pump

11. Will you still be collecting in 5 years' time?
    - ☐ Yes
    - ☐ No

12. Which factors are important to you in choosing a collection shore? (Please select up to 3)
    - ☐ Distance from parking
    - ☐ Distance from home
    - ☐ Distance from main road
    - ☐ Ease of access to the sand
    - ☐ Amount of lugworms present
    - ☐ Size of lugworms present
    - ☐ Type of sand (e.g. texture)

Periwinkle Collection Questionnaire

Date:

Location:

1. Which shores within North Northumberland do you collect periwinkles from?
   - ☐ Boulmer
   - ☐ Newton
   - ☐ Alnmouth
   - ☐ Howick
   - ☐ Beadnell
   - ☐ Craster
   - ☐ Seahouses
   - ☐ Burnmouth
   - ☐ Holy Island
   - ☐ Scremerston
   - ☐ Other:

2. Which other shores that you know of are collected in the area?
   - ☐ Boulmer
   - ☐ Newton
   - ☐ Alnmouth
   - ☐ Howick
   - ☐ Beadnell
   - ☐ Craster
   - ☐ Seahouses
   - ☐ Burnmouth
   - ☐ Holy Island
   - ☐ Scremerston
   - ☐ Other:

3. How often do you collect?
   - ☐ Daily
   - ☐ Every other day
   - ☐ Twice a week
   - ☐ Weekly
   - ☐ Fortnightly
   - ☐ Monthly
   - ☐ Every few months

4. How long per tide do you collect for?
   - ☐ Less than 1 hour
   - ☐ 1-2 hours
   - ☐ 2-3 hours

- ☐ 3-4 hours
- ☐ 4-5 hours
- ☐ More than 5 hours

5. How many periwinkles do you collect in this time (weight)?
   - ☐ 0-10kg
   - ☐ 10-20kg
   - ☐ 20-30kg
   - ☐ 30-40kg
   - ☐ 40-50kg
   - ☐ More than 50kg

   or

   - ☐ 0-5 pounds
   - ☐ 5-10 pounds
   - ☐ 10-20 pounds (1 stone)
   - ☐ 2-3 stone
   - ☐ 3-4 stone
   - ☐ 5-6 stone
   - ☐ 6-7 stone
   - ☐ More than 7 stone

6. What months do you collect in?
   - ☐ January
   - ☐ February
   - ☐ March
   - ☐ April
   - ☐ May
   - ☐ June
   - ☐ July
   - ☐ August
   - ☐ September
   - ☐ October
   - ☐ November
   - ☐ December

7. Which days do you collect?
   - ☐ Weekends
   - ☐ Bank holidays
   - ☐ Weekdays

8. Do you only collect at low tide?
   - ☐ Yes
   - ☐ No

9. Does the low tide height effect when you collect (e.g. spring or neap tide)?
   - ☐ Yes – I only collect at the lowest tide heights (e.g. spring tides)
   - ☐ No – I collect at any height of low tide
   - ☐ Yes/No – I try to collect at the lowest tide heights, but I also collect at higher low tide heights as needed

10. Do you collect in the dark?
   - ☐ Yes – But just early mornings and winter evenings
   - ☐ Yes – Including the middle of the night (after midnight and before 4am)
   - ☐ No – I only collect in daylight hours

11. Will you still be collecting in 5 years' time?
   - ☐ Yes
   - ☐ No

12. Which factors are important to you in choosing a collection shore? (Please select up to 3)
   - ☐ Distance from parking
   - ☐ Distance from home
   - ☐ Distance from main road
   - ☐ Ease of access to the rocks
   - ☐ Amount of periwinkles present
   - ☐ Size of periwinkles present
   - ☐ Type of rock (e.g. flat, boulders, pools)

**Appendix B: Aerial Imagery of Modelling Result Aspects**

Figure A:1: Budle Bay to Beadnell Bay – Aspect seen in Figures 3.4 and 3.5.

Figure A:2: Killiedraught Bay and Coldingham Bay – Aspect seen in Figure 3.6 A.

*Figure A:3: Holy Island, Fenham Flats, and Budle Bay – Aspect seen in Figure 3.6 B.*

*Figure A:4: BNNC EMS – Aspect seen in Figures 3.7 A, 3.8 A, and 3.9 A.*

Figure A:5: Berwick – Aspect seen in Figure 3.7 B.

*Figure A:6: Newton – Aspect seen in Figures 3.7 C and 3.9 D.*

Figure A:7: Boulmer – Aspect seen in Figures 3.7 D and 3.9 E.

*Figure A:8: Fenham Flats – Aspect seen in Figures 3.8 B and 3.9 B.*

Figure A:9: Budle Bay – Aspect seen in Figures 3.8 C and 3.9 C.