The University of Hull

Interaction between the Yorkshire coast static gear crustacean fishery and offshore wind energy development.

Being a Thesis submitted for the Degree of Doctor of Philosophy

In the University of Hull

By

Michael Roach

B.Sc. (Hons) Coastal Marine Biology (University of Hull)

October 2019

Abstract

Globally the offshore wind energy sector has seen an increase in the number of and spatial scale of offshore wind farms in the last decade. Offshore wind farms can be seen as many EU member states answer to meeting their energy demands from renewable sources. The increase in offshore wind developments can create spatial conflict with other marine users such as commercial fisheries. Their ecological effects on macro-benthic crustaceans are not currently widely understood. This thesis focuses on the short-term effects of the construction and operation of the Westermost Rough offshore wind farm and the subsequent closure and reopening of the site to fishing exploitation due to the construction process. There were limited effects of the Westermost Rough offshore wind farm on the size structure and catch rates of the commercially exploited crustaceans sampled over three survey years. The closure of the site during construction saw an increase in the size, abundance, and total egg yield of lobsters from the site. This increase in lobsters produced an adverse effect on the commercial bycatch species in the site. Reopening of the site to fishing exploitation, produced an immediate, short-term increase in effort. The increase in lobster size, abundance and total egg yield produced a dramatic decrease but within six weeks, reflected that of the control area. This thesis demonstrates that the there are few observable short-term effects of offshore wind farm construction on commercially exploited crustacean species. The thesis also demonstrates the effects of a closed area on commercial crustaceans and the effects of reopening the site to exploitation. The results can be used to assist in marine spatial management and future offshore wind interactions with commercially important crustacean fisheries.

Contents

List of figures

List of tables

List of equations

List of acronyms and abbreviations

ac – Automatic Count (generated from egg counter)
ANOSIM – Analysis of Similarities
ANOVA – Analysis of Variance
AW – Abdomen Width
BACI – Before/After, Control/Impact
BACI-PS – Before/After, Control/Impact – Paired Series
CCMAMLR – Convention for the Conservation of Antarctic Marine Living Resources
CEFAS – Centre for Environmental, Fisheries and Aquaculture Science
CFP – Common Fisheries Policy
CL – Carapace Length
CPUE – Catch per Unit of Effort
CW – Carapace Width
DEFRA – Department of Environmental, Food and Rural Affairs
DW – Dry Weight
ECDF – Empirical Cumulative Frequency Distribution
EFF – European Fisheries Fund
EMFF – European Maritime Fisheries Fund
EU – European Union
FAD – Fish Aggregation Device
FAO – Food and Agricultural Organization of the United Nations
FLOWW – Fisheries Liaison with Offshore Wind and Wet Renewables
GLMM – Generalised Linear Mixed Model
GW – Giga Watt
HFIG – Holderness Fishing Industry Group
HVDC – High Voltage Direct Current
IFCA – Inshore Fisheries and Conservation Authority (NE – North Eastern)
IUCN – International Union for Conservation of Nature
JNCC – Joint Nature Conservation Committee
kg – Kilogram
km – Kilometre
K-S – Kolmogorov Smirnov
K-W – Kruskal Wallis
LPUE – Landings per Unit of Effort
MCZ – Marine Conservation Zone
MDS – Multi-dimensional Scaling
MLS – Minimum Landing Size
MMO – Marine management Organisation
MPA – Marine Protected Area
MSY – Maximum Sustainable Yield
MW – Mega Watt
NFFO – National Federation for Fishermen's Organisations
nm – Nautical Miles
NOAA – National Oceanic and Atmosphere Administration
NTZ – No Take Zone
OWF – Offshore Wind Farm
P2 – number of eggs on the second pleopod
s.d. – Standard Deviation
SAGB – Shellfish Association of Great Britain
sd – Dry weight of the sub-sample of eggs
SFF – Scottish Fishermen's Federation
sn – Number of eggs in the sub-sample
SOM – Size at Onset of Maturity
S-W – Shapiro Wilkes
sw – Wet Weight of the sub-sample of eggs
TAC – Total Allowable Catch
tw – Total Wet Weight
WMR – Westermost Rough
WW – Wet Weight

Acknowledgements

Firstly, I would like to thank my supervisor, Dr Magnus Johnson, for the continued support, advice, patience, and continuous heated debate over the last six years. His guidance and support helped me in the academic process of finishing my thesis whilst balancing work commitments.

I would like to express my gratitude to the staff and members of the Holderness Fishing Industry Group for their support and resources made available to me throughout the PhD. Their advice and guidance on the local fishery have been integral to the understanding needed to complete my thesis. I would also like to thank Independent Shellfishermen's Co-operative (Bridlington) for introducing me to the lobster fishery in the early days of my work.

I would like to thank Ørsted for their use of the data gathered during the surveys and the guidance and support of Dr Andy Revill throughout the period.

Finally, I would like to thank my wife and family, their support throughout the last six years has been invaluable in this long journey to completing the thesis. I could not have done it without you.

Chapter 1: General Introduction

1.1 Lobster and crustacean fisheries

Crustacean fisheries have seen an increased growth over recent times, and their importance to rural fishing communities can be essential. Lobsters are one of the most economically important crustacean fisheries globally, for example *Homarus gammarus* first point of sale value can reach up to £24/kg (Marine Management Organisation (MMO), 2019). Lobster fisheries can be split into four main types: American lobster (*Homarus americanus*, H. Milne Edwards, 1837), European lobster (*Homarus gammarus*, L. 1758), rock lobsters (*Jasus* spp.) and spiny lobsters (*Panulirus* spp. and *Palinurus* spp.) (Pereira and Josupeit, 2017). There are also regional fisheries in some tropical regions for slipper lobsters (*Scyllaridae* spp.). The clawed lobsters (*Homarus* spp.), are targeted on both sides of the Atlantic, *H. americanus* on the eastern seaboard of the United States of America (USA) and *H. gammarus* in European waters (Figure 1.1).

Homarus spp. are targeted using static gear in the form of baited traps (USA) or pots (UK, subsequently referred to as 'pots'), the pots are deployed on the seabed for an immersion period and are designed to maximise retention of the target species. There are catches reported from other gear types such as static nets and bottom trawls, although they are generally bycatch (non-target species) as opposed to the target whitefish species (Lovewell, 1991). The technical advancement of pot fisheries has focused on increasing durability of the pot (constructed of metal as opposed to wood) and increasing their selectivity (bycatch reduction measures).

Globally, *H. americanus* landings have increased four-fold in the last 40 years, increasing from 38,447 t in 1979 to 162,547 t in 2017 (FAO, 2019). *H. gammarus* is landed in much smaller quantities globally due to their more limited range and lower abundance. Landings of *H. gammarus* have seen nearly a three-fold increase, increasing from 1739 tonnes in 1979 to 4713 tonnes in 2017 (FAO 2019). Lobsters are seen as a high value catch due to the high market value retained, both at first sale value (the value a fisher receives), between £9 - £24/kg for *H. gammarus* (MMO 2018), and the final consumer.

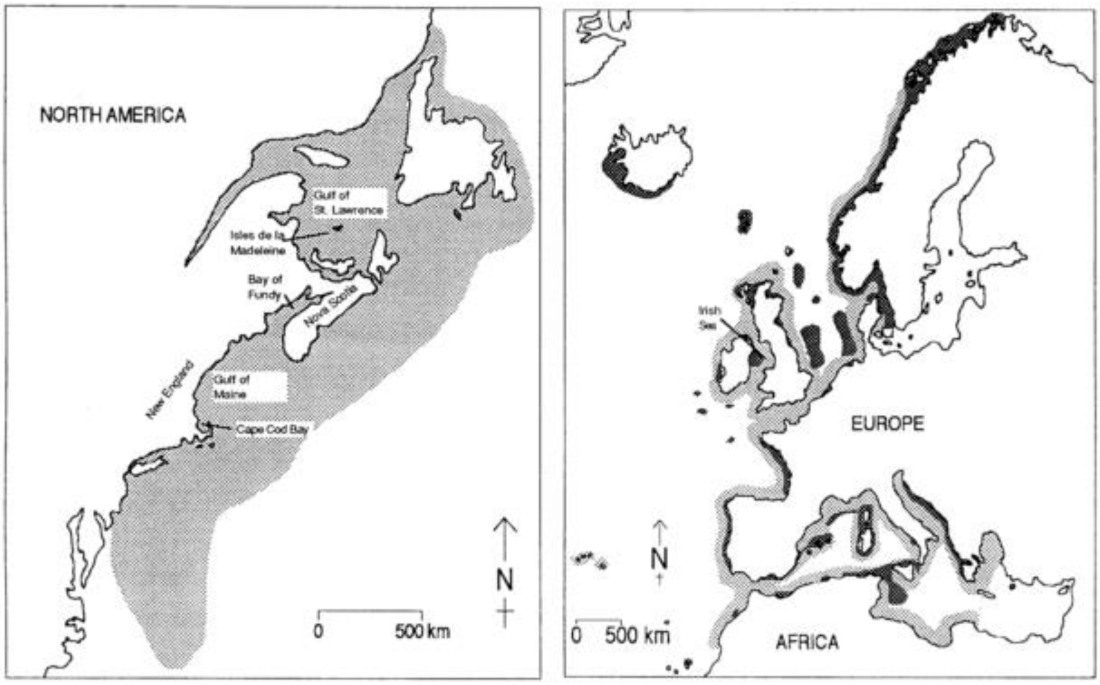


Figure 1.1: Range maps of the clawed lobsters, *Homarus americanus* in North America and *Homarus gammarus* (grey) and *Nephrops norvegicus* (black) in Europe and Northern Africa. Image taken from (Cobb and Wahle 1994).

In the USA, the Gulf of Maine fishery is the most important *H. americanus* fishery, accounting for 81.2% of the total *H. americanus* landed in 2017 (NOAA Fisheries, 2017). The UK is the European country with the most productive *H. gammarus* fisheries, exporting up to 80-85% of the catch to the European Union (EU) (Pereira and Josupeit, 2017). Bridlington: located on the north-east English coast referred to as the Holderness Coast, accounts for 24.4% of the total landings into England (MMO 2017b). The value of both fisheries, whilst different in scale, is economically important to the local communities and the additional sectors involved with the fisheries. It is estimated that for every fisher at sea, when locals are employed, there are a further 4 - 7 jobs created ashore (Merideth, 1999). The greater abundance of *H. americanus* entering the market generates a lower price per kg for the species (mean £8.77/kg in 2017 (State of Maine Department of Marine Resources, 2018)) in comparison to *H. gammarus* (mean £15.28/kg (MMO, 2019a). Whilst *H. americanus* accounts for 60% (in 2013) of global lobster landings (Pereira and Josupeit, 2017), their export into areas such as the EU can be dependent on *H. gammarus* landings. The main market for *H. gammarus* is the EU, with France, Italy and Spain accounting for the most imports. These countries show a preference for *H. gammarus* and imports of *H. americanus* increase when there is a shortfall of *H. gammarus* supply (Pereira and Josupeit, 2017).

The markets for *Homarus* spp. and many other lobster fisheries can fall into three categories: live animals, lobster tails and processed lobsters (cooked and frozen). The most valuable market is the live trade of lobsters where considerable infrastructure investment has been made on the Holderness Coast to ensure the viability and quality of the catch remains high through the chain to final consumer. This has included large storage tanks on the quayside and investment in vivier systems for transportation, both at sea and in the supply chain. However, this increase in live trade of animals outside of their natural range has increased the possibility of accidental or in some cases intentional introduction into overseas areas. In 2015, 361 *H. americanus* and 35 Dungeness crab (*Cancer magister*, Dana 1852) were released in UK waters by Buddhists as part of a religious ceremony incurring a £28,220 penalty (MMO 2017a). The pathways for introductions are not always clear, live, non-native lobsters are meant to be kept in separate tanks with no escape pathway(Jørstad *et al.*, 2011), however, there have been 26 reports of *H. americanus* in UK waters over the last 30 years (Stebbing *et al.*, 2012; Ellis *et al.*, 2017).

The increased global market for lobsters and other marine crustaceans may have encouraged increased levels of exploitation of target species beyond sustainable levels. High levels of exploitation can have a detrimental effect on fisheries targeting slow growing species. It is estimated that lobsters can take between 4 – 5 years to recruit into a fishery dependent on the prescribed minimum landing size (MLS) (Bannister and Addison, 1998). Continuous exploitation of commercial fisheries have been observed to create juvenescence (maturing at a smaller size) due to fishery induced selection which can cause instability in population dynamics (Anderson *et al.,* 2008). This loss of larger individuals in a population due to fishing mortality can potentially affect the overall reproductive potential of the population and future viability of a stock (Moland *et al.*, 2010; Gwinn *et al.*, 2015). The MLS is set at a size where the individual has the opportunity to breed at least once prior to being recruited into the fishery. This may cause an iteroparous (multiple reproductive events) population to shift towards a semelparous (single reproductive event) population, where there is only one chance of breeding prior to capture. Targeting of larger individuals within a stock, via length based management strategies, can also cause a truncating of the size distribution (Gwinn *et al.*, 2015). Although this is well documented in finfish fisheries, there is limited evidence of such for lobster fisheries. Melville-Smith and De Lestang (2006) reported a decline in the size at maturity of *P. cygnus* over time. Moland *et al.* (2013) observed a long-term decrease in natural mortality within a marine reserve, which could offset/reverse fishing induced selection within a *H. gammarus* population. However; Watson *et al.* (2017) observed that although lobsters were mating at a smaller size, ~28% were immature. They speculated that the immature females were mating post moult and using the protection that the larger males provided during the shelter guarding phase. The sexually active, immature females may lead to misrepresentation of juvenescence in a population. This can be further compounded by there being a lack of standardised methodology for determining size at onset of maturity estimates in crustaceans (Haig *et al.*, 2016).

In baited static gear fisheries, various bait sources are used from a variety of suppliers. In the USA it is predominantly herring (*Clupea harengus*, L. 1761), influencing catch rates and the quantity of herring used in the USA pot fisheries has been shown to improve growth rates of *H. americanus* in comparison to other bait sources (Saila *et al.*, 2002; Grabowski *et al.*, 2005, 2010; Harnish and Willison, 2009; Steneck and Wahle, 2013). Steneck *et al.* (2011) discussed the effect of artificially supporting a population via the introduction of mass quantities of bait, creating a 'gilded trap', simplifying the food chain for target species and potentially affecting the viability of a population when the population has become dependent on the bait source used in the fishery. In UK fisheries the preferred bait for lobsters is mackerel (*Scomber scombrus*, L. 1758), however instead of the mackerel fishery selling whole fish for pot bait, it tends to be what remains after the fish has been processed for human consumption. Bait supplies for pot fisheries can often be in short supply with alternative bait sources being required (de Rozarieux, 2014). The management of lobsters fisheries should take into account the effect on the fisheries that are used for bait supply (Grabowski *et al.*, 2005).

1.2 Management and Co-management

Management of pot fisheries are not as prescriptive as traditional finfish fisheries. In the UK, lobsters are currently not a quota species and not subjected to the landings obligation of the common fisheries policy (MMO 2019). As such, the International Council for the Exploration of the Sea (ICES) currently only offer advice for *Nephrops norvegicus* for crustacean fisheries in English waters, however, there is an ICES working group on the biology and life history of crabs (lobsters were added to this working group in 2013) (ICES 2016). Stock assessments in English waters are conducted by the Centre for Environment, Fisheries and Aquaculture Science (CEFAS). There are currently six stock units defined for lobster in English waters (Cefas, 2017). CEFAS stock assessments for *H. gammarus* and *Cancer pagurus* stocks in English waters are conducted using length cohort analysis due to the difficulties in determining age of crustacean species preventing traditional age-based cohort analysis (Smith and Addison, 2003). A maximum sustainable yield (MSY) level proxy is set (35% of the virgin spawner per recruit) with further reference limits set as 15% of the virgin spawner per recruit. North eastern stocks in English waters in the 2017 assessment were assessed as beyond the maximum reference point whilst southern stocks were below the maximum reference points (Cefas, 2017). Section 2.2.3discusses the management strategies used within the lobster fishery that is the subject of this thesis.

Co-management of fisheries stocks is the concept of managers and stakeholders working together to correctly manage a stock (Acheson and Taylor, 2001). Local knowledge of stock dynamics, the way a fishery operates and the effects of climate on a stock can be essential when aiming for co-management of a fishery (Close and Hall, 2006). Top-down stock management purely based on scientific data collection as opposed to including fisheries dependant data, can lead to a misunderstanding of the status of the stock (Hall and Close, 2007). Acceptance of stock assessments has often been met with scepticism by the fishing industry and there has been historic distrust among relevant stakeholders (Mackinson and Nottestad, 1998; Johannes *et al.*, 2008; Moller *et al.*, 2016). The use of fisher collated data can be a contentious subject in fisheries management (Zukowski *et al.*, 2011). In data-poor fisheries, fisher data can be used as a supplementary tool and provide a wider time scale than scientific studies (Boudreau and Worm, 2010; Moller *et al.*, 2016). Mackinson (2001) successfully used a 'fuzzy logic' model to combine quantitative and qualitative data to predict adult herring distributions and Boudreau and Worm (2010) used local knowledge to fill a data gap in the groundfish/lobster predator/prey interaction. Globally, there has been a paradigm shift in fisheries management to take a more holistic approach, including ecosystem management and consideration to socio-economic impacts rather than focusing on traditional stock management (Obura *et al.*, 2002).

Successful co-management has been demonstrated in several fisheries. For example: the south-west potting agreement in the UK, in operation since 1978 is an agreement between static and mobile gear types to de-conflict gear interaction. This was initially a voluntary agreement between the different sectors (Blyth *et al.*, 2002) and subsequently supported by regional byelaw legislation in 1998 (Devon & Severn IFCA, 2018). Co-managed, periodic or rotational closures are used in the Indo-pacific to successfully manage resources for certain taxa, these were predominantly reef fish and invertebrates such as Holothuroidea (Cohen and Alexander, 2013; Cohen and Foale, 2013a; Cohen *et al.*, 2013a). The use of co-managed closures, needs to be flexible in their approach to legislation to ensure the effectiveness (Gnanalingam and Hepburn, 2015).

1.3 The use of closed areas as a management tool

Protected areas (an area with some form of protective legislation) in the marine environment can take various forms and have different levels of legislation associated with them. Many protected areas fall under the umbrella term of 'Marine Protected Area'

(MPA). MPAs can be designated for multiple reasons such as habitat conservation, protection of biodiversity and to restrict exploitation of marine resources. MPAs are often thought of as being used to protect biodiversity via the restriction of fishing exploitation; however, Costello and Ballantine (2015) reported that 94% of designated MPAs allowed fishing exploitation within their defined boundaries. Protected areas that restrict fishing exploitation are commonly referred to as 'No Take Zones' (NTZ), these tend to be a more stringent form of protected area. In the UK, there are only three designated NTZs: Lundy, Flamborough Head and Lamlash Bay (JNCC, 2011). NTZs have been demonstrated to promote biodiversity, promote growth and abundance of the target taxa, counter fisheries-induced selection, increase larval production from the area and enhance spill-over effects into surrounding areas (Howarth *et al.* 2017; Davies *et al.* 2015; Hoskin *et al.* 2011; Huserbråten *et al.* 2013; Moland *et al.* 2013). The advantages of closed areas on stock enhancement are much debated as many benefits are difficult to ascertain. For example, the time scale of studies is often not sufficient to identify spill over effects (Moland *et al.* 2013; Smyth *et al.* 2015; Vandendriessche *et al.* 2015). Many MPAs are not of sufficient size to cover the home range and depth profile of the target species (Moland *et al.* 2011a; Moland *et al.* 2011b) or have not been designated for a sufficient period to enhance slow growing organisms (Gnanalingam and Hepburn, 2015). The implementation of protected areas and NTZs can often be met with resistance and scepticism from commercial fisheries (Kaiser, 2005). Commercial fisheries often face spatial restriction due to maritime spatial planning (Suddaby, 2002). Designation of protected areas, gear conflict, aggregate extraction, fossil fuel exploitation and offshore wind developments can affect the spatial distribution of commercial fisheries (Christie *et al.*, 2014). Offshore wind farms (OWF) have been highlighted as areas for co-location of the energy and fishing sectors (Hooper and Austen 2014; Hooper *et al.* 2015; Stelzenmüller *et al.* 2016). OWFs can act as a closed area either due to the presence of turbines preventing fishing exploitation or the developer not allowing fishing to take place at the site (Punt *et al.*, 2009; Krone *et al.*, 2017)T.

1.4 Offshore Wind Development and Commercial Fisheries

Offshore wind developments or sites vary spatially, and their effect on fisheries and the ecosystems in which they are installed are not widely studied. To date, the largest OWF globally is the Walney (and Walney extension), located in the Irish Sea, consisting of 189 turbines, generating 1.026 GW of power (Ørsted, 2019a). In comparison, the only OWF currently operational in US waters, the Block Island OWF only has 5 turbines generating 30 MW of power (Deep Water Wind, 2019). The physical presence of turbines in the marine environment can create spatial conflict with static gear fisheries and introduce a high risk for mobile gear fisheries within the OWF (Hooper *et al.*, 2015). Previous offshore developments, such as oil and gas rigs, have not been as extensive as offshore wind in their spatial requirements. This could potentially alter the way a fishery operates in an area. Spatial conflict between commercial fisheries and the expansion of the offshore wind energy sector, could lead to a loss of fishing grounds. Loss of fishing grounds could result in a potential reduction in fishing exploitation, thus having a positive ecological effect. OWF, by rendering part or all their near-field footprint inaccessible to fishing exploitation may act as a de-facto NTZ (Christie *et al.* 2014). Fish abundance was observed to increase in the Danish Horns Rev 1 OWF, 7 years post build of the site in the absence of bottom trawling at the site (Coates *et al.*, 2016). Exclusion of mobile gear from an OWF was reported to be responsible for increases in abundance and biodiversity (Coates *et al.*, 2014; Stenberg *et al.*, 2015). In some circumstances the turbines can act as fish aggregation devices (FAD), providing new and additional habitat for fish species (Lacroix and Pioch, 2011). Vandendriessche *et al.* (2015) reported that this association was of smaller sized fish. This association may only be a localised effect, where fish aggregate around individual turbines and migration between turbines can be dependent on the distance between them (van Hal *et al.*, 2017). Habitat loss due to monopile installation can be offset by the presence of the monopile providing habitat area up to 2.5 times that lost due to its placement, thus increasing net gain (Wilson and Elliott, 2009). Biodiversity of lower trophic level species has been demonstrated to increase due to the turbines providing additional settling surfaces (De Mesel *et al.*, 2015). Bivalve abundance has also been demonstrated to increase in the absence of bottom trawling within an OWF (Krone *et al.*, 2013a; Bergman *et al.*, 2014). Modelling of the effects of OWF development has also been shown to demonstrate a positive response to upper trophic level species (Raoux *et al.*, 2017). Atlantic Cod (*Gadus morhua*, L. 1758) have been observed to predate small crustaceans (*Pisidia longicornis*, L. 1767) associated with turbine foundations (Link *et al.*, 2009).

The positive ecological effects reported in association with OWF tend to focus on the introduction of a new hard substratum and habitat into the ecosystem. Construction of OWF on sandy substrata has demonstrated a positive ecological effect due to this introduction of hard substratum. However: these were at sites that were previously subjected to bottom trawling and positive effects reported are likely due to the absence of bottom trawling as opposed to the presence of OWFs (Coates *et al.* 2014; De Backer *et al.* 2014; Hooper and Austen 2014; Stenberg *et al.* 2015; Vandendriessche *et al.* 2015). The effects of construction of OWF on different substrata is not currently well understood due to majority of OWFs being constructed on sand-based substrates. In particular, the effects of introducing hard substrate to ecosystems already characterised as such (cobble/rock etc.) is not reported, however, consideration is given in OWF planning processes to ensure minimum disturbance to key habitat features (English *et al.*, 2017). Addition of scour stone protection on the base of turbines may provide additional habitat for shelter dwelling organisms (Wilson and Elliott, 2009; Wilson *et al.*, 2010). However the association of fish species with turbines (Stenberg *et al.*, 2015; Coates *et al.*, 2016) may increase predator/prey interactions and cause a shift in the dynamics of the ecosystem.

1.5 Outline of thesis

1.5.1 Aim of the thesis

The principle aim of the thesis was to understand the impact of the Westermost Rough OWF development on the ecology of commercially exploited lobster fisheries and associated by-catch in the area. The secondary aim is to understand the effects of closing an OWF site to fishing exploitation during construction on the ecology and egg production of the lobster population and the ecology of the associated by-catch in the area.

1.5.2 Individual chapter aims

This thesis is separated into nine chapters, Chapters 1 and 2 focus on the literature and description of the fishery, Chapters 3 and 4 focus on the quantitative field results collected at the WMR OWF. Chapter 5 discusses a methodology that was needed to predict egg yield which was the focus of Chapter 6. Chapter 7 discusses the overall findings, conclusions drawn, and suggestions for further study. Chapter 8 is the literature presented in the thesis and supporting information and data are presented in the Chapter 9 – appendices.

- Chapter 1 presents a literature review, focussing on crustacean fisheries, management strategies and the growth of OWF developments and their effects on the marine environment. The chapter is divided into four subject headings: a description of lobster and crustacean fisheries, the concept of co-management and management of fisheries, protected areas in the marine environment and the development of OWF and their ecological effects. The aim of the chapter is to understand the relevant literature and current studies of relevance to the thesis.

- Chapter 2 is split into two parts: Part 1 aims to describe the fishery that is the subject of the thesis and Part 2 aims to describe the at sea sampling protocols for Chapters 3, 4 and 6.
- The aim of Chapter 3 is to understand the effects of OWF construction on the ecology of a commercially important lobster fishery and associated by-catch over three survey seasons.
- The aim of Chapter 4 is to understand the effects of exclusion of fishing effort from an OWF site during the construction period on the ecology of a commercially important lobster fishery and associated bycatch and the effects of subsequent re-opening of the site and assess if an OWF site can act as a protected area.
- The aim of Chapter 5 is to compare commonly used methodologies for determining egg number in *H. gammarus* and create a suitable model for determining egg number.
- The aim of Chapter 6 is to understand the effects of closure of an OWF site to fishing exploitation, due to construction, has on the egg production and density of ovigerous female lobsters in the site.
- In the final chapter of the thesis, Chapter 7, the results are summarised, and the key findings discussed. Limitations of the study are highlighted and suggestions for management and further work made. Final conclusions of the study are presented.

The researched published from this thesis is including in Chapter 9, Appendices and discusses the lobster data from Chapters 3 and 4.

Chapter 2: Background and description of the fishery, site description and sampling methods

2.1 Abstract

The Holderness Coast supports the largest lobster fishery in England, accounting for 24% of national landings. The evolution of the fishery is discussed, from a traditional whitefish fishery deploying mobile gear types to a predominantly static gear fishery targeting macrobenthic crustaceans. The development of the Holderness Fishing Industry Group is discussed and how it relates to their approach towards engaging with scientific research. The at-sea sampling methodology for subsequent chapters is described with regards to location, gear type used, and offshore protocols implemented.

2.2 2.2 Description of the fishery

2.2.1 Background of the fishery

The Holderness Coast fishery in east Yorkshire extends from Flamborough Head to Spurn Point at the mouth of the Humber estuary (Figure 2.1). Traditionally, the fishery targeted *Homarus gammarus* nearshore and inshore (inside 12 nm) in the summer months and *Cancer pagurus* offshore (outside 12 nm) in the winter months. Whilst this pattern still exists, in recent times this separation in fisheries seasonally is less distinct, with both species being targeted all year around. The main landing port in the region is Bridlington, with smaller ports in Hornsea and Withernsea (Figure 2.1). Additionally, beach-launched vessels also sail from Tunstall, Flamborough and Grimsby. The MMO separates the fleet, divided by overall length into < 10 m length (n = 12) and > 10 m length (n = 10) for management and recording requirements (MMO, 2019). For this study, these are vessels that have their home port registered as Bridlington, Hornsea or Withernsea with the regulatory authorities. Vessel numbers of members of the Holderness Fishing Industry Group (HFIG) differ from the official numbers: < 10 m length (n = 31) and > 10 m length (n = 23) (Jamie Robertson, pers. comm. CEO of HFIG). Grimsby vessels were excluded as most do not fish within the Holderness fishery.

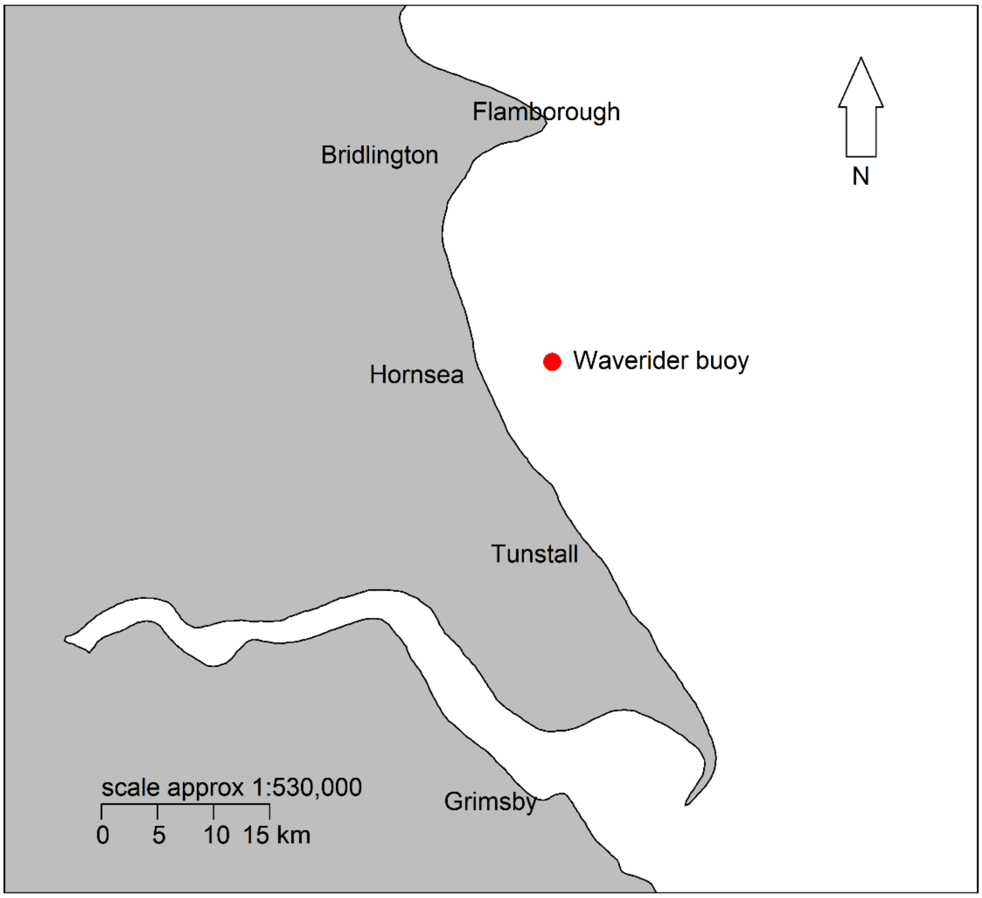


Figure 2.1: The Holderness coastline with the landing ports for the fishery ports labelled. Location of Hornsea Waverider (WMO ID: 6201019) denoted in red, position 53° 55'00 N, 000° 04.00' W.

The fleet was historically a trawling fleet, targeting whitefish such as Cod (*Gadus morhua*) and Haddock (*Melanogrammus aeglefinus*). Landings of whitefish into Bridlington saw a steady increase during the 1970's and reached its peak in 1985 (7538 tonnes) (MMO, 2019a). Subsequently there was a rapid decline in whitefish landings (with the exception of a single peak in 1993), to just 2 tonnes in 2016 (Figure 2.2 a & b)). Conversely, over the same period, landings of lobster and crab have increased steadily, reaching a peak of 453 tonnes of lobster in 2017 (Figure 2.2a) and 2507 tonnes of crab in 2016 (Figure 2.2b). Effort within the fishery has increased over the last 30 years, seeing a peak in total days fished in 2007 (n = 6537) in comparison to 1989 (n = 1449) (Figure 2.2c) (MMO, 2019a).

The switch in effort from whitefish to shellfish can be attributed to several factors. Stocks of whitefish into Bridlington saw a large decline in recent times (Figure 2.2a). Thurstan *et al.* (2010)reported an estimated 94% reduction in in UK demersal fisheries landings per unit of fishing power over the last 118 years. This does not reflect the increase in days fished by vessels targeting shellfish (Figure 2.2c) which have seen an increase in recent years. The variability of fuel costs can play an important part of a fisher's decision to alter their fishing practises. Fish prices have not risen to meet this increase in costs to the fisher causing a decline in overall profit per trip (Arnason, 2007; Abernethy *et al.*, 2010). Static gear fisheries are estimated to have fewer total costs associated with fuel than mobile gear, however the cost of fuel per unit of catch can be higher in static gear fisheries (Abernethy *et al.*, 2010).

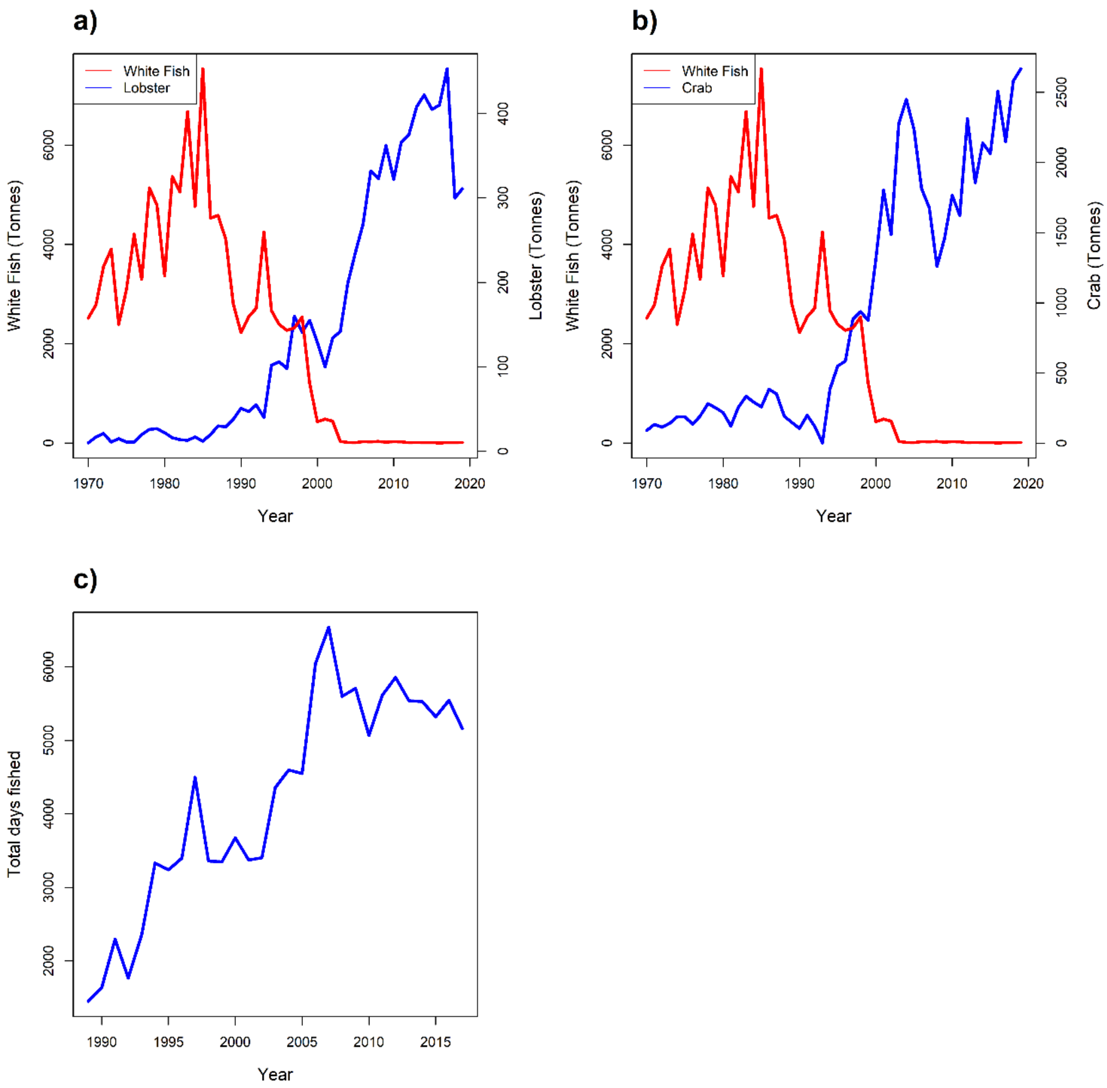


Figure 2.2: Landings of whitefish (left-hand axis) and lobster (a) and crab species (b) (right hand axis) into the port of Bridlington 1970 -2016. Data sourced from the Marine Management Organisation (MMO, 1970 - 2019). The total number of days fished for vessels targeting predominantly shellfish for the ports of Bridlington, Withernsea and Hornsea, 1987 – 2017 (c). Data sourced from CEFAS/HFIG fisheries science partnership 2018/19 – note difference in x-axis scale, these data are only from 1988 – 2017 as opposed to 1970 – 2017 in figure a & c.

The absence of predatory whitefish species such as Atlantic Cod (*Gadus morhua*) has been predicted to cause in an increase in abundance of crustacean species (Boudreau and Worm, 2010). There is potential for the reduced population size of an extensively overfished population becoming a permanent state, this has been observed in several finfish populations (Lees *et al.*, 2006), reducing the impact of predator populations on their prey species. Boudreau and Worm (2010) observed an inverse relationship between abundance of groundfish species and *H. americanus* abundance and *G. morhua* was also perceived by fishers to be the most influential predator of *H. americanus*. The reduction of predation on larval stage and juvenile lobsters can increase the chance of survival to adulthood and recruitment into the fishery (Link *et al.*, 2009). However, Hanson and Lanteigne (2000) dissected the stomach contents of *G. morhua* in the eastern Atlantic during pre and post collapse of the *G. morhua* stock in the region, and observed that there were *H. americanus* in the stomachs of only 1 out of 22,625 cod examined pre-collapse, and 6 out of 12,008 examined post collapse. Boudreau and Worm (2010) also observed a small proportion of lobsters observed in the stomachs of *G. morhua* (0.31%). Therefore, it is unlikely that the reduction in biomass of *G. morhua* led to an increase in lobster abundance in the region. In the absence of *G. morhua*, the competition for benthic resources may be reduced, allowing for an increased lobster population.

National legislation and the introduction of the EU Common Fisheries Policy (CFP) in the 1970's, the introduction of Total Allowable Catch (TAC) and quota allocation (the process by which a fisher is limited to a set quantity for different target species), may have also encouraged fishers to diversify away from whitefish (Markus, 2010). As part of the EU CFP in 1984 (European Parliament, 2019), the UK government was encouraged to reduce the capacity of the UK fishing fleet. The UK government contributed £53 million to reduce the demersal fleet by 17% (gross registered tonnage) (Holland *et al.*, 1999). This was done using several decommissioning schemes, where owners received grants for complete removal of their vessels from the fishery (Holland *et al.*, 1999). The positive effects of government subsidies and buyback schemes have been questioned, the schemes have been suggested to encourage a race to fish and unsustainable fishing practices (Markus, 2010; Cisneros-Montemayor *et al.*, 2016; Holland *et al.*, 2017). Many vessel owners also sold their quota, mainly to producer organisations, and it was estimated that by 1986 most of the UK demersal quota was held by producer organisations (Holland *et al.*, 1999). This reduction in the trawl fleet, saw a reinvestment in smaller more efficient vessels, without quota that may have forced the switch to alternative target species. Leocádio *et al.*, (2012) reported that potting for *Nephrops norvegicus* was potentially more economically viable than trawling. Thus, cumulative factors affecting profitability of a fishery may have been responsible for the shift from whitefish to shellfish in the Holderness region.

2.2.2 Current fishery

The Holderness coast fishery as of 2019 consisted of around 65 vessels that target almost exclusively shellfish, the target species were European Lobster (*Homarus* *gammarus*), Edible Crab (*Cancer pagurus*), Velvet Swimming Crab (*Necora puber*) and the Common Whelk (*Buccinum undatum*). The fishery uses static gear types in the form of baited creels (referred locally to as pots), the pot type used in the region is the parlour pot (Figure 2.3, right-hand pot) (Seafish, 2019a). The pot is designed to have one or more entrances to allow target species to enter the pot when attracted to the 'smell' of the bait. The entrances are designed in a way to make it difficult for the catch to escape, (Atema *et al.*, 1979) although it is still possible for the target species to escape via the entrances. The capture efficiency of a pot can differ between pot types and individual fishers, affecting retention rates and pot selectivity (Smolowitz, 1978). Retention rates of captured lobsters are thought to be low, for *H. americanus*, as little as 6% in the field (Jury *et al.*, 2001) or 11% in laboratory experiments (Karnofsky and Price, 1989). Fishermen often have a personal design for the entrances to their pots, aiming to increase retention rates. The technological modification in pot design over recent years has seen a move away from 'three boned' wooden pots to 3-5 boned steel pots with plastic coating which increases their durability and efficiency (as seen in Figure 2.3 right-hand pot). Improvements in pot design have been demonstrated to enhance catch rates in lobster and crab fisheries (Lovewell *et al.*, 1988). Additionally, there has been technological improvements in the way pots are deployed and recovered. Hydraulic haulers for recovering pots from the seabed have replaced hauling pots by hand and many UK vessels operate self-hauling systems (Jamie Robertson, pers. comm, CEO of HFIG).

Pots are set in a string of 10 – 80 pots, the number of which is generally determined by the size of the vessel operating them, and vessels have to be able to carry a string when recovering prior to redeployment. The strings are then deployed on suitable ground for a set period. This immersion period is referred to as the soak time and varies depending on the fisher, season, weather and target species (Bennett, 1974). Bennet & Lovewell (1977) observed that when soak time was less than 5 days the variability of catch did not differ significantly when compared to soak times > 5 days. Retention rate of pots can vary depending on their design (Jury and Watson, 2013). Jury *et al.*, (2001) observed that in the *H. americanus* fishery in New Hampshire, USA, the retention rate for lobsters was just 6% of the total that interacted with the pot. Brčić *et al.*, (2018) reported a probability of 50% retention rates for pots targeting *N. norvegicus* greater than 31.69 mm carapace length using a 40 mm mesh size, this was 59% over the Minimum Landing Size (MLS) for the species.

Figure 2.3: Typical parlour pot used in the Holderness fishery and the fine mesh pot used in the survey to capture smaller animals. The fine mesh pot (left) has a 30 mm mesh and base length of 762 mm, the parlour pot (right) has a 70 mm mesh and 965.2 mm base length.

The success of pot fisheries can depend on the bait used and the quantity. Harnish and Willison (2009) estimated for the Nova Scotia lobster fishery, 1.9 units of bait were used for every unit of lobster caught, this increased to 3.0 in peak season. The Holderness fishery uses several fish species to bait the pots. The most commonly used bait is the ‘frames’ (carcass left after filleting) of mackerel (*Scomber scombrus*) to target lobsters and the heads of salmon (*Salmo salar*) to target crabs. Whilst this is a by-product of other fisheries it holds a commercial value when sold as bait to pot fisheries. The cost of bait to the fisher is estimated to be 10-11% of turnover and upwards of 7532 tonnes of bait are used in the UK annually (de Rozarieux, 2014). The use of the waste/bycatch from other fisheries, reintroduces marine protein into the marine environments as opposed to on-land disposal. Baited pot fisheries can actively feed target species, increasing their chance of survival, and supporting their growth. This can be more prevalent in larger lobsters, smaller lobsters have been observed to have more natural diet compositions (Grabowski *et al.*, 2005, 2010). Steneck *et al.*, (2011) discussed the effect of the ‘gilded trap’, a process of artificially supporting a population and encouraging a monoculture using baited pots. This was observed by Jury *et al.*, (2001), whilst studying pot retention rates, the only species observed entering the pots was *H. americanus*, indicating a bias towards this species in the study area. This is not observed in the Holderness fishery, diversity of animals associating with pots on the ground is much greater, the results of which are discussed in subsequent Chapters 3 and 4.

Effort on each of the target species in the Holderness fishery is seasonally dependent. Edible crabs are targeted throughout the year and the main landings are from summer to winter (Figure 2.4, (MMO, 2019a)). Lobster landings by weight are significantly lower than edible crab by volume and their main landing period is over the summer months (Figure 2.4). Over the past 5 years there has been a temporal shift in the lobster landings, with an increase in the number of lobsters landed in the earlier months (Figure 2.4). This can be attributed to milder winters in recent years resulting in higher sea surface temperatures (CEFAS, 2018). Mean sea surface temperature for the months November to March, increased 1ºC (s.d. +/- 1.55) between 2013 and 2017 (CEFAS, 2018). Mobility of lobsters has been linked to sea surface temperature, with greater movement observed as temperature increases (Smith *et al.*, 1999; Moland *et al.*, 2011c).

Lobsters demonstrate a seasonal migration to deeper water during cooler months, tracking the warmer waters offshore (Moland *et al.*, 2011a), thus the milder winters may have brought forward the migration inshore. As lobsters migrate offshore, effort is reduced on the stock due to the offshore grounds being restricted to vessels that can operate offshore.

The migration of lobster's results in the fishing effort change to edible crabs during the cooler months. During the warmer months, the lobsters migrate inshore and generally undergo ecdysis for breeding. This period is referred to colloquially as "new shelling" and is when effort is greatest, involving targeting the lobsters that have recently moulted and re-hardened their shell.

Since 2015 peak lobster landings have demonstrated a peak in the summer months, however overall landings have reduced since 2018 (Figure 2.4). The reduction in monthly lobster landings since 2018 can be attributed to the ban on landing ovigerous lobsters being introduced in October 2017 (DEFRA, 2017). Monthly lobster landings increase steadily as sea surface temperature increases (Figure 2.5). At 14ºC there is a large increase in landings (x 1.3 increase from 13ºC), indicating this is the optimal temperature for targeting lobsters. This needs to be considered with the milder temperatures being associated with more suitable conditions for vessels being able to fish.

Temperature is one of the governing factors affecting metabolic rate in lobsters and driving their feeding strategies (Mente *et al.*, 2001). Warmer temperatures increase metabolic rate and thus feeding activities and the probability of encountering a baited pot and subsequently being caught. The temperature in the southern North Sea is increasing at a faster rate than the surrounding areas due to the effects of climate change (Belkin, 2009).

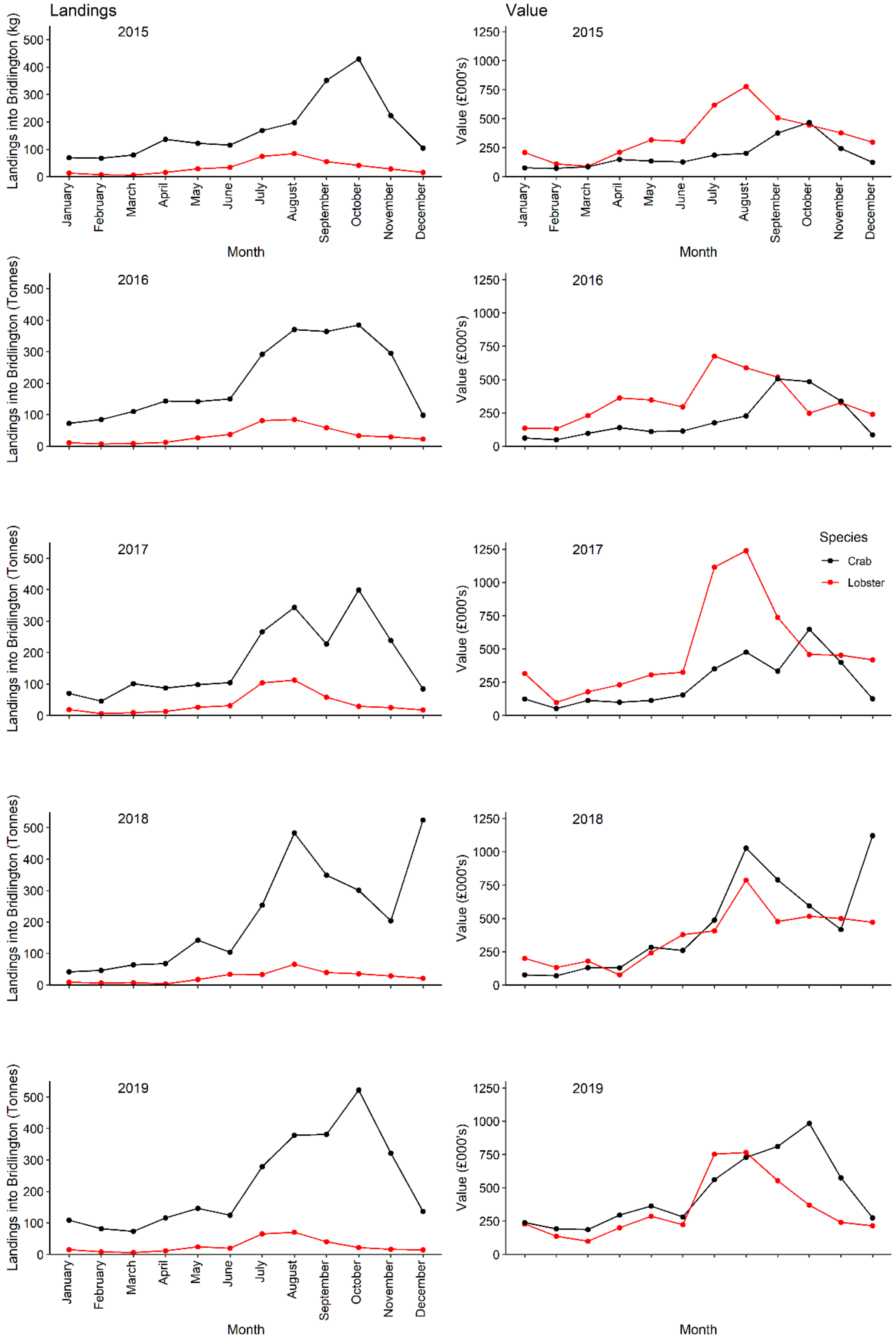


Figure 2.4: Monthly landings statistics from 2013-17, for edible crab and lobster (left hand plots) and their respective value (right hand plots). Data derived from MMO landing statistics (MMO, 2011 - 2017).

This temperature increase could increase the yield in a lobster fishery. However, temperature increases beyond optimal levels can be detrimental. For example, larval release can occur at suboptimal times and development can be affected due to increased temperatures (Schmalenbach and Franke, 2010; Small *et al.*, 2016). Many *H. americanus* fisheries have seen a decline in landings due to the effects of warmer temperatures being beyond the limit of the species tolerance and affecting fecundity and recruitment (Mills *et al.*, 2013; Steneck and Wahle, 2013; Koopman *et al.*, 2015; Le Bris *et al.*, 2018), although this may benefit fisheries further north, where temperatures are lower.

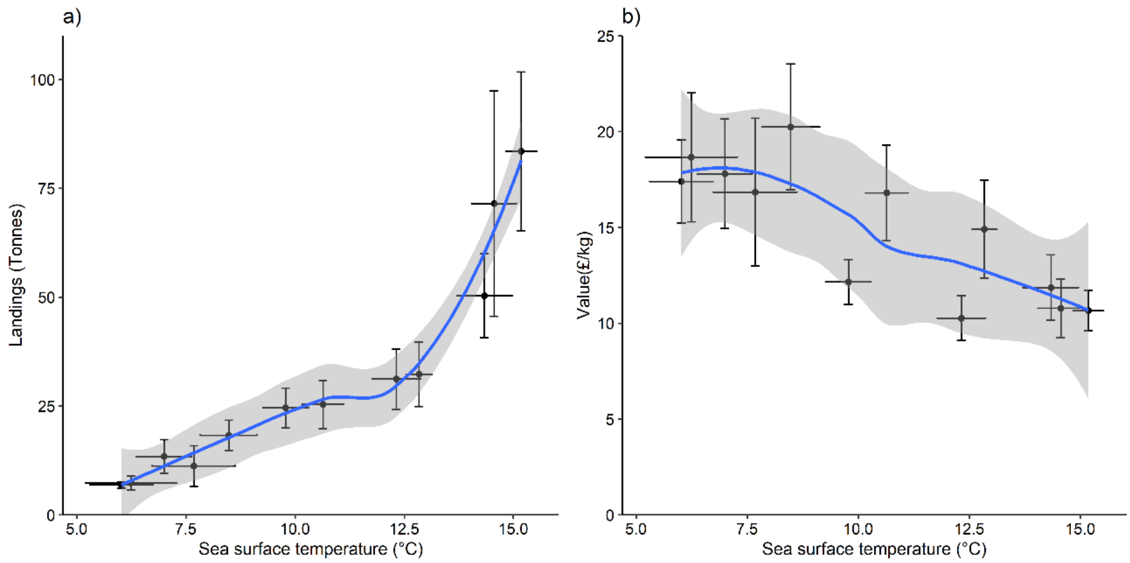


Figure 2.5: Mean monthly landings for lobster (a) and mean value of lobster per kg (b) (x axis) and mean monthly sea surface temperature (y axis) from 2015 – 2019. The error bars (x and y) illustrate the standard deviation of the mean and the grey shaded represents the 95% confidence intervals. Sea surface temperature taken from the Hornsea wave rider buoy (53º 55.00' N, 000º 04.00'E), part of the CEFAS WaveNet (CEFAS, 2018). Landings and value data derived from MMO landings statistics for Bridlington (MMO, 2019).

Although lobsters have a lower tonnage landed than edible crab, they have a higher price per/kg (Figure 2.6). Lobster first-sale value varies seasonally from £8-24 per kilo, whereas historically edible crab value only changes over the season by < £0.50 per kilo (2014 – 2019) (Figure 2.6). However in 2019, mean crab prices increased by 22%, from £1.69 to £2.17 per/kg, this can be attributed to opening markets in Asia (MMO, 2018) The total value of lobster landings in 2019 was £4.07 million in comparison to £5.49 million for edible crab. This first sale value is estimated to significantly contribute to the local economy. There are estimated to be 389 fishers operating in the region of the Yorkshire coast (Scarborough port of administration) MMO, 2019a), and based on the estimates of Merideth (1999) (4 -7 jobs ashore for every fisher at sea), this could contribute between 1556 – 2723 jobs on shore.

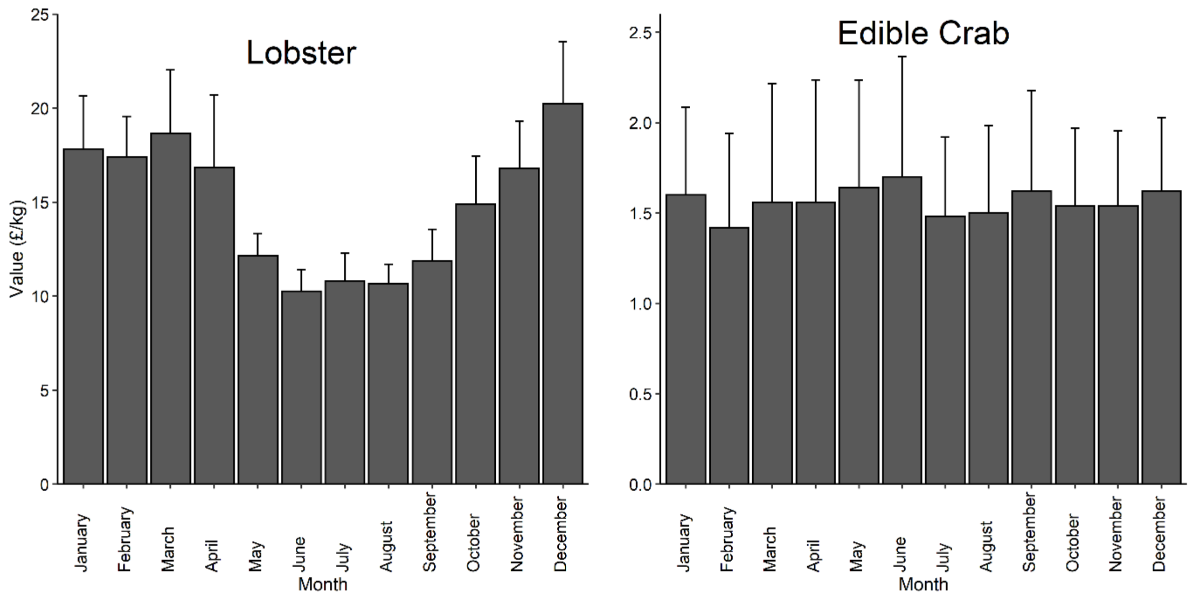


Figure 2.6: Monthly average value per kilogram for lobster and edible crab from –2015 - 2019. The top of the bars represents the mean value per kilogram and the top of the error bars the standard deviation of the mean (MMO, 2019a).

2.2.3 Legislation

Many crustacean species are not subject to quota legislation. Lobsters, edible and velvet swimming crabs are non-quota species and are controlled nationally and on a regional scale. National legislation prevents the landing of ovigerous edible crabs and in October 2017 there was national legislation introduced to prevent landing of ovigerous lobsters (DEFRA, 2017). Shellfish are often legislated regionally by a minimum landing size (MLS) preventing landing of juvenile catch. The Holderness fishery falls under the regional jurisdiction of the North Eastern Inshore and Conservation Authority (NEIFCA (0-6 nm)) and the Marine Management Organisation (MMO (6-12 nm)). The shellfish stocks are legally controlled by the Crustacea Conservation byelaw (NEIFCA, 2015). The main legislation is the minimum landing size (MLS) for each species: lobster 87 mm carapace length (CL) and edible crab 140 mm carapace width (CW), (increased from 130 mm CW in 2016), and the equivalent MLS for velvet swimming crab is 65 mm CW. There are also prohibitions against landing “nones” (lobsters missing both claws) and lobsters with mutilated tails, to aid in stock conservation via protecting low market value lobsters (NEIFCA 2015). In 2016 NEIFCA also introduced mandatory escape gaps in all pots within their district (up to 6 nm from the baseline), this mandated the addition of an escape gape measuring 80 mm x 466 mm in both the baited and parlour end of the pots. In addition, there has long been a voluntary “V-notch” programme in the fishery. This is a process by which a fisher cuts a “V” out of the telson of an undesired (generally low quality) or breeding female lobster (Figure 2.7), this lobster is then protected via regional legislation until the ‘V’ has grown out to a depth of 5 mm or less. This can take up to two years. V-notching has been used in recent times to protect ovigerous females during the brood period and is commonly used in many lobster fisheries (Acheson and Gardner, 2011). There is the possibility that fishers can V-notch a lobster for selfish reasons. For example, if a lobster has a soft shell and holds little market value, a fisher may V-notch it to prevent another fisher landing it once the shell re-hardens (Dale Rodmell, pers. comm. Assistant Chief Executive, NFFO). This however aids in the conservation effort irrespective of the motives behind the V-notch.

Figure 2.7: Dorsal view of *H. gammarus* telson, highlighting a V-notch cut into two uropods for conservation purposes. Fishers often cut a V-notch in two sections to increase visibility of the conservation method.

2.2.4 Formation of the Holderness Fishing Industry Group

Many regional fisheries that target similar species can form fishermen's organisations, this can also be driven by economic factors. Cooperatives can form, to act as merchants, buyers and sellers for the catch and also lenders for vessel investment (Yvonne Webb, pers. comm. Director, HFIG). In the Holderness region there were informal organisations of fishers such as the Bridlington and Flamborough Fisherman's Association. There were several developments affecting the fishery that required exclusion zones and caused disruption and displacement in the fishery such as the installation of the Langaled and York gas pipelines that make landfall at Easington (Suddaby, 2002). Initially fishermen coalesced around key individuals to discuss the developments with the aid of scientific advice. Subsequently, development of the Westermost Rough (WMR) offshore wind farm was proposed. The approach of the wind farm developers did not fully consider the complexities of disruption and displacement of fishing effort (Rodmell and Johnson, 2002; Suddaby, 2002). In 2011 a more structured fishermen's organisation was formed to represent the whole fishery: the Holderness Fishing Industry Group (HFIG). HFIG appointed a CEO who was not a fisherman and came from a legal and marine biology background. This was due to the fishermen needing representation with regards to the development of offshore wind energy in the region, who could understand and effectively liaise with both developers and fishermen. HFIG has a board of directors that represent the key elements within the fishery including representatives from the inshore/offshore and beach launched fleet, all the merchants and the largest shellfish processor in the region, as well as employed staff (Figure 2.8). The use of influential members from the capture sector of the fishery as board members ensures that the fishery is represented from all aspects of the capture sector, ensuring information is disseminated quickly via their daily communication with the members.

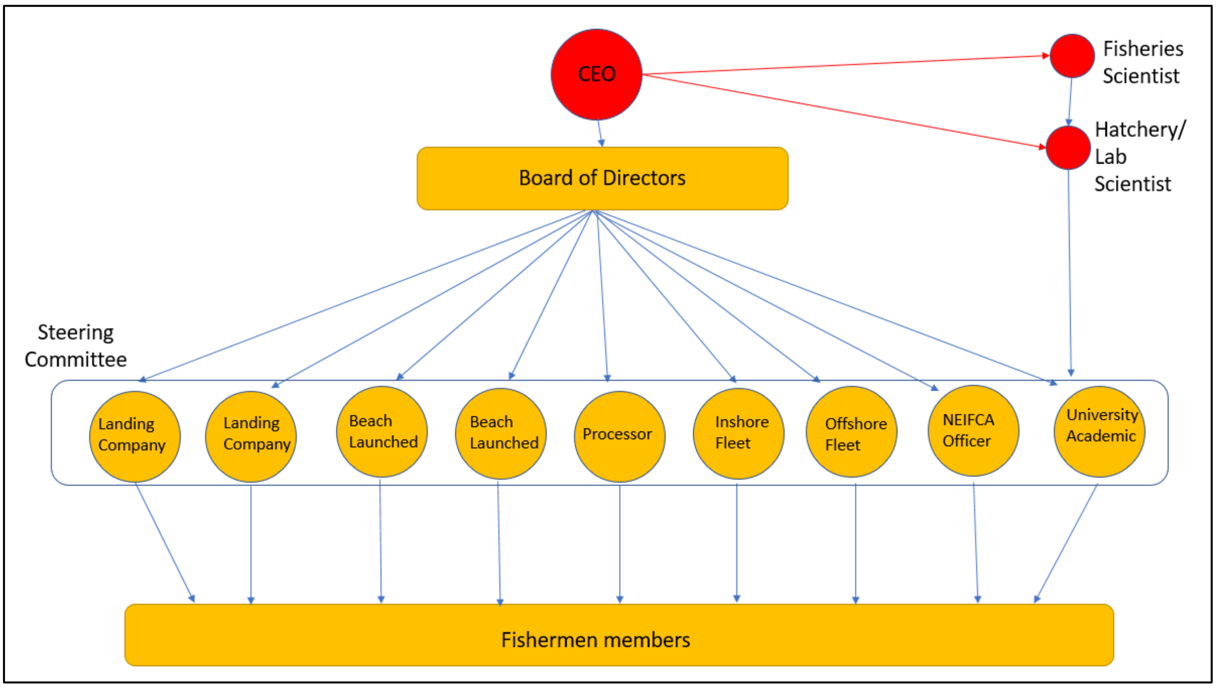


Figure 2.8: Schematic of the structure of HFIG staff (red) and the board of directors, steering committee and fishermen members (yellow).

The consultation process with regards to the planned construction of the WMR OWF was the main catalyst for the formation and cohesion of HFIG. The appointment of a CEO meant the fishery could speak to the developer with one voice rather than the skipper/owners of every vessel representing itself. This resulted in a unique discourse between the developer and the fishery, leading to the developer appointing the fishery to conduct the impact assessment of the WMR development on their stocks. Since then, HFIG has also acted as a representative in discussions with government agencies (NEIFCA, MMO, CEFAS, DEFRA), aided in grant applications for the fishers to European Fisheries Fund (EFF) and European Maritime and Fisheries Fund (EMFF). HFIG has also conducted outreach to local communities and NGOs describing the fishery and the local species and has conducted scientific research with academic institutions.

In 2012, HFIG purchased a research vessel, the *R.V. Huntress* (Figure 2.9), to conduct, amongst other projects, the impact assessment of the WMR wind farm construction. This formed part of the HFIG's goals, to conduct scientific research on the fishery, to collate and evaluate data for any marine spatial planning conflicts. The *R.V. Huntress* was crewed by two ex-fishermen and had a fisheries scientist/observer employed by HFIG to conduct the research. Scientific studies, specifically impact assessments, are often met with skepticism by the fishing industry (Suddaby, 2002; Hooper and Austen, 2014b; Hooper *et al.*, 2015). This is often because scientific studies do not/cannot reflect the way that a fisherman would operate and there can be a lack of understanding of the scientific process by the fishing industry. This is not the case of all fishers and industry data collection/collaboration can enhance the traditional, empirical approach (Johnson and Van Densen, 2007; Hoare *et al.*, 2011). The use of a fishing industry owned, operated and crewed research vessel, deploying and operating scientific surveys that are robustly designed has negated many of these concerns in the fishery and further enhances the understanding of the scientific process by the fishing industry.

Figure 2.9: *R.V. Huntress*, research vessel owned and operated by HFIG.

2.3 Study site description and general sampling methodology and protocols

2.3.1 Site description

The study site for the evaluation of the effects of wind farm construction and the effect of closed areas (Chapters 3, 4 and 6 respectively) was the Westermost Rough OWF. Samples were also collected for Chapter 5 from the control site. The specific methodology is described in each chapter.

The Westermost Rough OWF is located within the Holderness fishery area, on the north-east coast of England (Figure 2.10). The wind farm consists of 35, 6 MW turbines and associated sub-station and is approximately 35 km^2. The wind farm extends from 7.7 km off the coast to 13.3 km. The substrate is predominantly rock and cobble with sand patches (Titan Environmental Surveys Limited, 2013). The WMR OWF was one of the first to be constructed on this type of habitat. Prior to the construction of the site in 2014/15, the site was subjected to boulder removal throughout. The depth range in the area is 15 to 23 m.

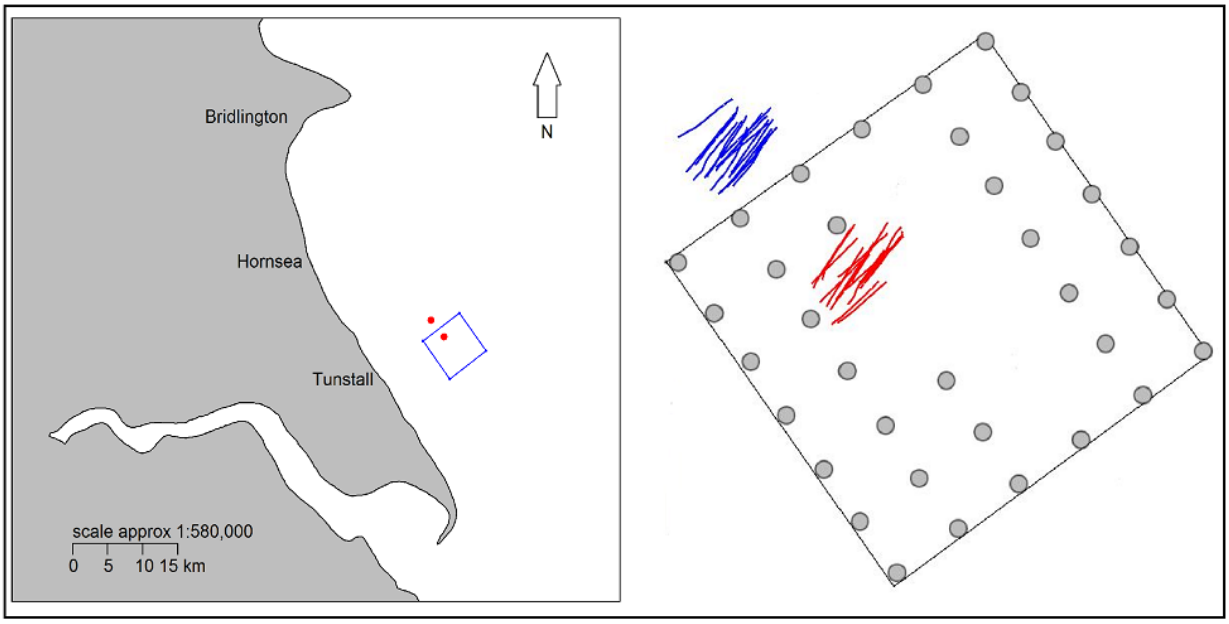


Figure 2.10: Location of the WMR OWF, the individual turbines marked and the locations of the control strings of pots to the North of the site and impact strings within the turbine array. The strings represent the sites where they are typically deployed, these strings are from the 2015 survey.

2.3.2 At sea sampling methods

The at sea sampling methodology was designed by a steering committee consisting of HFIG staff and fishermen members, NEIFCA Scientific Officer, Fisheries Liaison Officer for Ørsted (OWF developer), an independent scientist and an independent fisheries observer. Sites were selected based on fisheries experience in the area and the limitations of impact site selection was due to Ørsted requirements for sites not to impact on the construction of the OWF. There has been criticism of control sites being used for comparison to treatments when the two sites do not have similar features and habitats (Lindeboom *et al.*, 2015). The control site, whilst only 1 km from the impact sites, was identified due to the site having a similar depth profile, distance from shore and substrate type (Titan Environmental Surveys Limited, 2013). Additionally, the prevailing residual current in the area is north to south, thus any effects of the wind farm should not be observed to the north of the site.

The intention behind the survey design was to emulate the before/after, control/impact (BACI) approach (Lindeboom *et al.*, 2011, 2015). However, due to limitations in both the sites available and the opportunities to gather baseline data (only 2013 available) it was decided to undertake a before/after, control/impact – paired series (BACI-PS) approach (Franco *et al.*, 2015: Thiault, *et al*., 2017), with the OWF sites as the impact and the control site to the north as the control.

Pre-construction surveys were conducted in 2013, post construction surveys were conducted in in 2015 and 2017. Strings of shellfish pots (Figure 2.11**Error! Reference source not found.**) were deployed at the locations shown in Figure 2.10, the impact site was the turbine array and the control site 1 km to the north of the site.

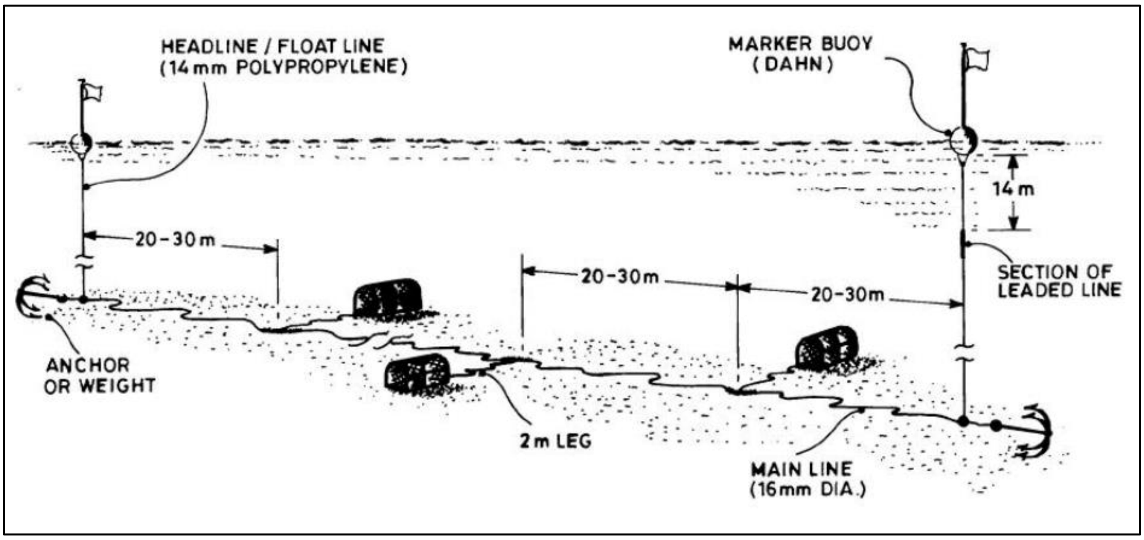


Figure 2.11: Schematic of the construction of a string of lobster pots deployed on the seabed. In the Holderness fishery, marker dans are replaced by inflatable buoys. Distances and specifics of the string construction in this survey are detailed above. Image provided from the Seafish database (Seafish, 2019b).

The strings of pots were deployed to reflect the commercial fishing effort in the area. Commercial fishing effort in the area use parlour pots to catch both lobsters and crab species, however lobsters are the dominant species during the survey periods. The abundance of lobsters captured in a pot can affect the abundance of crab species captured (Bennett, 1974; Addison, 1995), demonstrating an inverse relationship between lobster and crab species capture rates within pots (Figure 2.12). Therefore, all other species were defined as commercial bycatch (crab species) or non-commercial bycatch (other invertebrate catch and fish species).

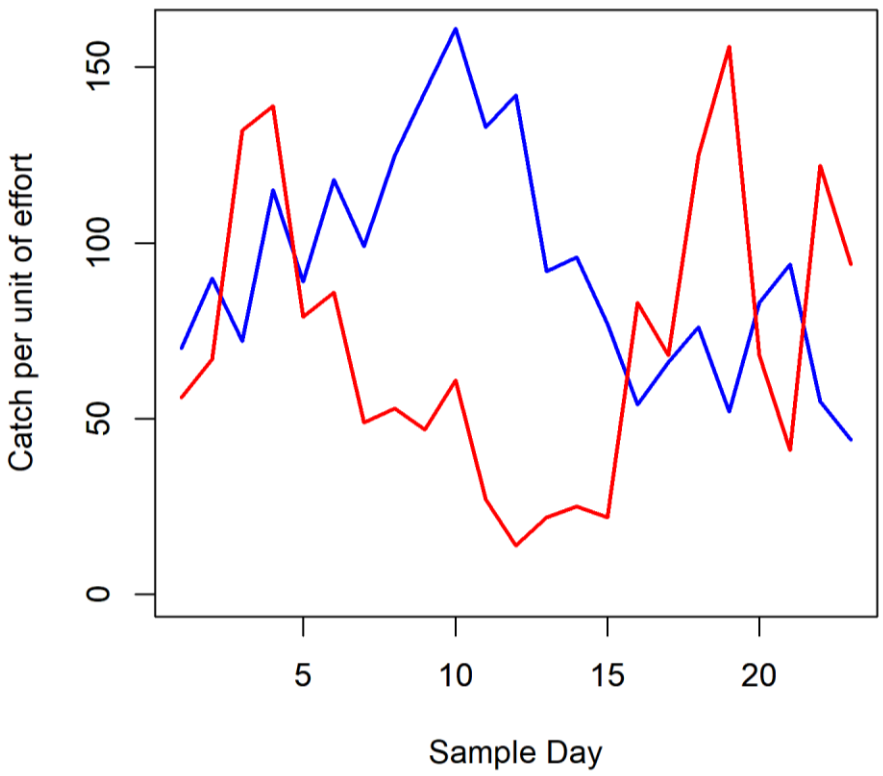


Figure 2.12: Catch per unit of effort of lobsters and edible crabs per sample day, demonstrating inverse relationship between the catch rates of the two species. Data sourced from HFIG.

Each sampling string consisted of 30 parlour pots, 25 pots (96.5 cm base and 70 mm mesh (Figure 2.3 right-hand pot) and 5 fine mesh “prawn pots” (76.2 cm base and 30 mm mesh) (Figure 2.3 left-hand pot). The fine mesh pots are typically used in *N. norvegicus* creel fisheries. Their deployment in this survey was to retain the smaller lobsters and commercial bycatch that may escape the larger mesh pots, ensuring an accurate representation of the lobster and commercial bycatch size structure and catch rates in the survey sites. The fine mesh pots were deployed every sixth pot in the string. Each pot was 40 m apart in the string with a 2 m leg from the main rope to the pot. The string was anchored at either end with a 20 kg anchor and marked with a surface marker buoy (Figure 2.11). The rope used was 13.5 mm in diameter and was leaded (weighted with lead woven into the fibre) to reduce movement in currents. The overall length of each string was 1100 m and they were deployed, as is the norm for this region, in a north/east, south/west orientation.

Within the fishery, fishermen have individual preferences for bait used (up to 12 different species), for example salmon (*Salmo salar*) heads are used to target edible crabs. However, mackerel (*Scomber scombrus*) is predominantly used to target lobsters and is the most available and commonly used bait. It was decided to use bait that targeted lobsters due to their high economic value and lobsters being dominant in the area during the survey period. All pots were baited with mackerel frames of the same quantity (2 frames, approximately 0.2 - 0.3 kg) and quality during the entire sampling period.

Pots were subjected to a soak time (immersion period) of approximately three days for the summer months (June – September inclusive). After a three-day immersion period, a single string of 30 pots each was hauled at both the impact and control sites and individuals of all taxa recorded per pot. The pots were rebaited and redeployed at the site for the next immersion period. No sub-sampling was conducted on the catch and all individuals of lobster and commercial catch were recorded All catch was returned to the sea post sampling.

The individuals sampled were catagorised as commercial catch (lobster), commercial bycatch (edible crab and velvet swimming crab) and non-commercial bycatch (all non-commercial species) including fish species such as cod and dab (*Limanda limanda*). These were deemed as bycatch rather than commercial due to few fishers in the region having quota allocations to land fish species.

Due to difficulties in recording all required data at the pot level (spatial limitations on the vessel and safety restrictions), it was decided that the abundance of all individuals was recorded within each pot for each site, allowing for catch per unit of effort analysis. At each site, on hauling the string, the different species were separated and segregated by sex then processed for biometric data once the string had been rebaited and relocated on the survey site. Size of animal was determined as carapace length (CL) for lobsters, measured from the sub-orbital spine to the posterior of the carapace. Carapace width (CW) was used to determined size of edible crab and velvet swimming crab and was recorded at the widest point of the carapace. Size of individuals was measured to the nearest mm. Sex and ovigerous (egg bearing) status was recorded for each individual. A semi-qualitative condition index was created (Table 2.1) and applied to each commercial animal, this was based on the market value of the catch allowing for landings per unit of effort analysis. Abundance of non-commercial bycatch was recorded for each pot and total length of all fish species was recorded.

Table 2.1: Condition index developed by HFIG to determine the quality of the animals captured. The index was based on factors affecting the market value of the catch.

Grade	Description
1	Healthy specimen, no physical impairments or signs of disease or biofouling
2	Soft shelled (pre or newly moulted)
3	Missing pereopods but not chelipeds, no visible signs of disease or biofouling
4	Missing chelipeds and possibly pereopods, no visible signs of disease or biofouling
5	Visible signs of disease e.g. black spot or large amounts of biofouling (> 10%), but no physical impairments
6	Visible signs of disease e.g. black spot or large amounts of biofouling (> 10%), but physical impairments and missing chelipeds and pereopods or possibly both.

Data analysis specific to each chapter is included within the methodology section of individual chapters.

Chapter 3: A three-year study of the effects of offshore wind farm construction and operation on the ecology of commercially important crustacean fisheries

3.1 Abstract

Offshore wind farms are an important component of the energy production and the strategic response to the threat of climate change for many countries. There is potential for conflict between their development and other marine users such as commercial fisheries. Static gear fisheries targeting important commercial shellfish species can be particularly affected by spatial restrictions due to offshore wind farm developments. The short-term effects of the construction of the Westermost Rough offshore wind farm on the commercially important *Homarus gammarus* and associated *Cancer pagurus* and *Necora puber* commercial bycatch are investigated. The study follows the paired series approach assessing the ecological impacts on the size structure and catch statistics of the fishery. *H. gammarus* saw an increase in size and catch rates over the three years, followed by *N. puber*, both of the species were influenced by a variety of factors including the wind farm construction. Bycatch diversity differed over the three years but not between sites within each year. Further study of the site will highlight any long-term effects of the operational phase of the offshore wind farm.

3.2 Introduction

There has been worldwide increase in energy provided from renewable sources such as wind energy, surpassing 63 GW in 2015, an 18% increase since 2014 (Global Wind Energy Council, 2015). Offshore wind energy developments are often the most used tools by national governments to meet both their energy demands and their commitment to renewable energy sources. For example, within Europe there is a commitment for member states to obtain 10% of their energy from renewable sources by 2020 to meet EU targets of 20% of energy generated from renewable sources by 2020 (European Commission, 2016). The UK is one of the forerunners in developing offshore wind facilities, with 25 sites currently operational and a further 16 with consent for development (The Crown Estate, 2017).

Research into the interactions between OWFs and the marine environment has shown a steady increase in recent years. A Google Scholar search for “offshore wind” and “marine” between 1998 – 2008 highlighted 4210 articles whereas between 2008 and 2018 there were 17,900 articles, a threefold increase in the literature. However this increase has been largely review-based with few empirical studies available (many of which form part of unpublished studies in the form of environmental impact assessments), this makes it difficult to ascertain a reliable assessment of the cumulative impacts of offshore wind energy developments on the marine environment (Lindeboom *et al.*, 2015). The published literature has largely focussed on interactions between offshore wind developments and seabirds (15 out of 78 publications reviewed by Hooper *et al.*, (2017)), marine mammals (Bailey *et al.*, 2010; Brandt *et al.*, 2011; Madsen *et al.*, 2006; Thomsen *et al.*, 2008), substrate and infaunal disturbance (De Backer *et al.*, 2014; Vandendriessche *et al.*, 2015; Coates *et al.*, 2016),fish populations (Bergman *et al.*, 2014; De Troch *et al.*, 2013; Stenberg *et al.*, 2015; Wahlberg & Westerberg, 2005). The potential habitat enhancement and subsequent benefits have also been discussed (Wilson and Elliott, 2009; Krone *et al.*, 2013a, 2017; Kamermans *et al.*, 2018; Sas *et al.*, 2018; Coolen *et al.*, 2019; Tonk and Rozemeijer, 2019). There is a need for research into key areas such as: turbidity effects of the monopiles, facilitation of invasive species migration (“stepping stone effect”) and noise and vibration on the benthos (Dannheim *et al.*, 2019a).

The construction of an OWF has the potential to change the habitat and alter the ecosystem within an area (Wilson and Elliott, 2009; Wilson *et al.*, 2010). The addition of individual monopiles/turbines and their associated sub-stations into the marine environment introduces an array of artificial hard substrata into an area. This can provide new surfaces for colonising species such as bivalves and algae (Hooper and Austen, 2014b) and additional habitats in the form of scour stone protection, which is located to protect the base of the monopiles/turbines (Wilson and Elliott, 2009). New settling surfaces in the form of monopiles and associated scour stone protection can provide habitat for colonisation by lower trophic level species (De Mesel *et al.*, 2015). However, the species that colonise the new substrate may not necessarily reflect the surrounding biota, predominantly due to the new substratum being of different physical properties to the surrounding substrate (Krone *et al.*, 2013). There is concern that the increase in wind farms may act as a mechanism for invasive species colonising an area – the turbines acting as "stepping-stones" for colonisation (Dannheim *et al.*, 2019a).

The introduction of both the monopiles/turbines and associated scour stone protection has been demonstrated to increase biodiversity and biomass of associated fauna in some OWF sites (Lindeboom *et al.*, 2011; Krone *et al.*, 2013a, 2013b, 2017; De Backer *et al.*, 2014; Stenberg *et al.*, 2015; Tonk and Rozemeijer, 2019). For example, the associated scour stone protection has been demonstrated to support large numbers of *C. pagurus.* Krone *et al.*, (2017) observed 5000/m^2 juvenile *C. pagurus* on individual monopiles which was more than double that of monopiles without the scour stone protection. This was predicted to contribute to 27% of the *C. pagurus* population production.

To date most of OWF constructed in European waters are typically in less than 30m water depth and constructed on sand-based substrata (Roach *et al.*, 2018). The habitat enrichment effects of OWF built on substrata that are already characterised by hard substrate (e.g. cobble/rock) is yet to be understood. Comparison of studies from different OWF located on different substrata can lead to a misunderstanding of the effect of their construction and subsequent operation (Lindeboom *et al.*, 2015).

OWF are often located in areas that conflict with commercial fisheries (Coates *et al.*, 2016). The physical presence of turbines can deter or exclude certain fishing activities (Hooper and Austen, 2014b; Hooper *et al.*, 2015). Mobile fishing techniques are the most affected by the presence of OWF as it can be impractical or unsafe to tow mobile gear through an OWF. This absence of fishing exploitation has been demonstrated to have a positive effect on biodiversity in some areas (Bergman *et al.*, 2014; Coates *et al.*, 2016). Static fishing may still co-exist with OWF if the operator allows fishing within their site. In an OWF that has excluded fishers, Stelzenmüller *et al.* (2016) assessed that for a static netting fishery there would be a loss of ~50% of earnings, however this was not the case for the static pot fishery in the same study. There has been focus on the co-location of certain gear types and fisheries but these have focussed on static pot fisheries (Hooper and Austen, 2014b; Stelzenmüller *et al.*, 2016) and OWF as potential aquaculture sites

(Griffin *et al.*, 2015). Mobile gear studies have largely focused on the benefits of mobile gear not having access to OWF sites as opposed to the potential of co-location (Bergman *et al.*, 2014; Coates *et al.*, 2014, 2016).

3.2.1 Aims, hypotheses and objectives

The aim of Chapter 3 is to understand the effects of OWF construction on a commercially important lobster fishery and associated by-catch over three survey seasons.

The hypotheses to be tested in Chapter 3 were:

3.1. The size structure and catch rates of lobsters and associated commercial bycatch will be affected by the construction and operational phases of the Westermost Rough offshore wind farm development when compared to a control site over a three-year period.

3.2. The community assemblages derived from pot bycatch will be affected by the construction and operation phases of the Westermost Rough offshore wind farm development.

The objectives of Chapter 3 were:

- Data on size structure and catch rates of lobsters and associated bycatch will be collated via at sea sampling during a pre-construction survey (2013) and subsequent post construction surveys (2015 & 2017) at the Westermost rough offshore wind farm development and associated control site.
- Comparisons of size structure and catch rates of lobsters and associated commercial bycatch will be made in each year between the wind farm and control site and also interactions between years will be investigated.
- Comparisons of community assemblages derived from non-commercial bycatch caught in pots will be made in each year between the wind farm and control site and also interactions between years will be investigated.

3.3 Methods

At sea surveys were conducted in accordance with the protocols described in the methodology section (Section 2.3.2). Data was gathered during at sea sampling days during the summer months of 2013 (pre-construction), 2015 and 2017 (post-construction) at both the impact site of the Westermost Rough OWF and associated control site (Figure 2.10). Comparisons were made between the OWF and associated control site following a before/after, control/impact - paired series (BACI-PS) approach (Franco *et al.*, 2015: Thiault, *et al*., 2017), Comparisons of size structure and catch rates of lobsters and commercial by-catch were made between the two sites, across all three survey years (2013, 2015 & 2017) to test the null hypotheses: '*Size structure and catch rates of lobsters and associated commercial by-catch did not differ significantly between the OWF and control over the three survey years'*. Community assemblages derived from all commercial and non-commercial bycatch were compared between the wind farm and control site over the three survey years, testing the null hypotheses '*Species richness, abundance, diversity and community assemblage did not differ significantly between the wind farm and control site over the 3 survey years*'.

3.3.1 Data analysis

All analysis for Chapter 3 was conducted using R statistical software (R Core Team, 2017) except for multi-dimensional scaling (MDS) and Analysis of Similarities (ANOSIM) analysis conducted using PRIMER v6 (Clarke & Gorley 2006). Generalised linear mixed effect model (GLMM) was applied using the lme4 package in R statistical software (Bates *et al.*, 2015). Packages ggplot2 (Wickham, 2009), gridExtra (Auguie, 2017), dplyr (Wickham *et al.*, 2017), car (Fox and Weisberg, 2011), Matching (Sekhon, 2011) were used for data manipulation, analysis and graphical outputs.

3.3.1.1 Size distribution

Kolmogorov-Smirnov (K_S) two-sample test can be used to compare the length frequency distribution of two samples (Ogle, 2016a). Differences in length frequency of lobsters and commercial bycatch for 2015 and 2017 were compared to the pre-construction data in 2013 and analysed using a two-sample K-S test. Results of the K-S test were represented graphically using Empirical cumulative frequency distribution (ECDF) plots (Ogle, 2016). Generalised linear mixed effect model (GLMM) can be used when the length frequency data are not normally distributed and when there is potential for pseudo-replication (Zuur & Ieno, 2016). The survey design introduced the potential of pseudo-replication by sampling the same sites every 3 days for the summer period in each survey year. The length frequency data of both the lobsters and commercial bycatch did not conform to a normal distribution at either the impact or control site over the three survey years (Shapiro Wilkes (S-W), $p < 0.05$). GLMM was deemed as the most suitable analysis. Therefore, a GLMM was applied in which the relative catch probability of lobsters and commercial bycatch of each size entering the pots within each year was the response variable, carapace length/width was the fixed effect and haul (survey day) was the crossed random intercept. A binomial error was applied due to the response variable being the relative catch probability of commercial catch entering pots within each year. Sex, condition and ovigerous status were investigated as fixed effects and discounted from the GLMM as non-significant variables (sex & condition, $p > 0.05$) or bias towards female catch (ovigerous status). Soak time was investigated to assess whether it should be accounted for within the GLMM, however there was a poor relationship between daily abundance of commercial catch and soak time ($r^2 < 0.1$ on all occasions). This was further negated by the survey design as both control and wind farm sites were hauled on each occasion and subjected to the same soak times. Linear, cubic, quadratic and constant models were generated and trialled to best describe the relationship. There was no significant difference in the linear, cubic and quadratic models generated (Analysis of Variance (ANOVA), $p > 0.05$). Therefore, the simplest model was the best description of the relative commercial catch probability of each size entering the pots between the baseline year in 2013 and the subsequent years 2015 and 2017.

$$\Pr\{\frac{Test}{Test + Control}\} = 1/(1 + e^{-(\text{haul} + \beta_1 \times \text{length} + \beta_2 \times \text{length}^2)}) \quad (1)$$

This follows similar methodology described by Holst & Revill (2009), analysing difference in catch composition at length between control and treatment experiment trawls (Van Marlen, *et al.*, 2014; Vogel *et al.*, 2017). Within the GLMM (Equation 1), **Test** was determined as the survey year post build for both 2015 and 2017. In all cases **Control** was determined as the baseline data set in 2013. There were equal number of survey days in 2013/15 (n = 23), however there were only 16 survey days in 2017. Therefore, for the GLMM analysis only, the data were sub-sampled between the 2013/17 size data, selecting survey days from 2013 that reflected as close as possible the same sampling days as 2017.

Validation of each GLMM was conducted by checking that the standardised residuals conformed to a normal distribution (Shapiro-Wilks (S-W), $p > 0.05$, in all cases) (Thomas *et al.*, 2015) and also the results were compared to the two sample Kolmogorov Smirnov test. GLMM results are presented graphically, allowing inference as to where the difference lay within the distribution.

3.3.1.2 Catch comparison

Catch per unit of effort (CPUE) was determined as the total number of each commercial species caught in a string of pots (n = 30) (Davies *et al.*, 2015). The total number of commercial species in each string that were above their respective MLS and of good quality was determined as the landings per unit of effort (LPUE) i.e. the number of individuals that a fisher would land to market. The CPUE and LPUE data conformed to a normal distribution (S-W, $p > 0.05$) however the variances could not be considered equal (F-test, $p < 0.05$). Due to the difficulty of conducting non-parametric analysis for repeated measures (same sites over three years), potential for type II error and the data conforming to normality, it was decided to conduct a two-way, repeated measures analysis of variance (ANOVA). This was to analyse if the CPUE and LPUE of lobsters and commercial bycatch differed significantly between the impact (OWF) and control site and between the years 2013, 2015 and 2017 and if there was a significant interaction between site and survey year. Due to non-homogeneity of variances, validation of the ANOVA models was conducted by checking the standardised residuals conformed to a normal distribution (S-W, $p > 0.05$, in all cases) (Thomas *et al.*, 2015).

3.3.1.3 Non-commercial Bycatch diversity

Non-commercial bycatch was defined as an individual of any species that was not targeted intentionally that holds no market value to the fishery, inclusive of the catch of all invertebrate species. This included commercial species such as lobster, edible and velvet swimming crabs that were below their respective minimum landing size (MLS) or above MLS but of poor quality. *Gadus morhua* (Atlantic cod) were also deemed as bycatch (even above MLS) as most of the local fleet do not hold quota for whitefish.

Species richness (S), total individual abundance (n) and Shannon-Weiner (H’, LOGe) diversity indices were calculated using PRIMER v6 (Clarke & Gorley 2006), and analysis assessing differences in S, H’ and n between the wind farm and control site and the three survey years was conducted using R statistical software (R Core Team, 2017). Species richness (S) and Shannon-Weiner diversity indices (H’) did not conform to a normal distribution (S-W, $p < 0.05$), therefore a Kruskall-Wallis (K-W) test was applied to analyse if S or H’ differed significantly between survey years and a Wilcoxon rank sum test was applied to analyse S or H’ differed significantly between sites within each survey year. Replicates were determined as a string hauled on each survey day. Total number of individuals (LOGn) conformed to a normal distribution (S-W, $p > 0.05$) and could be considered equal (F-test, $p > 0.05$), therefore a two-way repeated measures ANOVA was applied to analyse if LOGn differed significantly between years/sites and if there was a significant interaction between site and year.

Differences in community assemblage between the impact and control site and also between survey years was investigated. Multi-dimensional scaling (MDS) was applied to a Bray-Curtis similarity matrix of each of the test variables to present graphically, a similarity clustering of the diversity indices of the non-commercial bycatch. Each MDS plot was generated 999 times and the best representation presented determined by the stress level. Stress levels below 0.1 were deemed as excellent representations and stress levels greater than 0.20 were deemed as unsuitable representations of the data (Clarke & Gorley 2006). Analysis of similarity (ANOSIM) was conducted on all test variables (999 permutations) to statistically analyse the difference in similarity of replicates within each survey year and similarity between sites.

3.4 Results

Over the three survey years a total of 35,469 commercial shellfish were recorded: 9,854 lobsters, 16,215 edible crabs and 9,400 velvet swimming crabs. Lobsters were recorded in greatest abundance in 2015 (n = 4619), followed by 2013 (n = 3108) and 2017 (n = 2127). Edible crab total abundance was greatest in 2013 (n = 9268) followed by 2015 (n =3744) and 2017 (n = 3203). Velvet swimming crab total abundance was greatest in 2017 (n = 4794), followed by 2013 (n = 2787) and 2015 (n = 1819). A presence and absence list of all non-commercial bycatch is provided in Appendices, Table 9.3.. Soak time varied between the three survey years. The discrepancy of soak times was due to inclement weather conditions and reflects a general feature of pot fishing. The shortest mean soak time was in 2013 (3.0 days, s.d. +/- 1.34), followed by 2015 (3.9 days, s.d. /- 2.1) and 2017 (4.1 days, s.d. +/- 1.5 days). There were 23 survey days conducted in 2013 and 2015 but 16 survey days conducted in 2017. This was due factors outside of control such as adverse weather conditions and mechanical faults with the *R.V. Huntress* during August 2017.

Descriptive statistics and supporting data for Chapter 3 analysis are presented in Chapter 9: Appendices.

3.4.1 Size distribution

Analysis was conducted comparing the length frequency distribution of the lobster and commercial bycatch at the impact site (OWF) and control site between the pre-construction survey (2013) and to the first-year post build (2015) and also at the impact site (OWF) and control site between the pre-construction survey (2013) and the third year post build survey (2017) .

3.4.2 2013/2015 comparison

3.4.2.1 Lobsters

The length frequency distributions of lobsters sampled in the wind farm was significantly greater in 2015 than in 2013 (K-S, $p < 0.001$, Table 3.1). The wind farm in 2015 showed a higher proportion of lobsters at a larger size (>100 mm CL) than sampled in 2013 (Figure 3.1a), there was a greater proportion of lobsters from the MLS of 87 mm to 96 mm CL sampled in 2013. There was a greater size range, 39 – 126 mm CL in 2015 as opposed to 56 – 114 mm CL in 2013. The Empirical Cumulative Distribution Function (ECDF) plot (Figure 3.1g) demonstrates that the greatest differences in distributions were between 75-92 mm CL. This was supported by the GLMM plot (Figure 3.1a), which demonstrates that there was a greater proportion of lobsters sampled over 70 mm CL in 2015 than in 2013. The size frequency distributions of lobsters sampled in the control site differed significantly between the two years (K-S, $p < 0.001$, Table 3.1). The control site in 2015 showed a larger proportion of lobsters below the MLS of 87 mm CL than in 2013. Larger lobsters sampled in the control site were observed in greater proportions in 2013 than in 2015 (Figure 3.1d). Figure 3.1d, demonstrates that in 2015 across both sites there was a greater distribution of lobsters than in 2013 across the size range.

Table 3.1: Results of K-S analysis of the size frequency distributions between years at the wind farm and control sites of the WMR OWF.

Species	Treatment	Site	Test Statistic	p
Lobster	2013<2015	Wind farm	0.10	< 0.001
Lobster	2013<2015	Control	0.21	< 0.001
Lobster	2013<2017	Wind farm	0.11	< 0.001
Lobster	2013<2017	Control	0.12	< 0.001
Edible Crab	2013>2015	Wind farm	0.19	< 0.001
Edible Crab	2013>2015	Control	0.16	< 0.001
Edible Crab	2013>2017	Wind farm	0.07	< 0.001
Edible Crab	2013>2017	Control	0.09	< 0.001
Velvet Swimming Crab	2013>2015	Wind farm	0.36	< 0.001
Velvet Swimming Crab	2013>2015	Control	0.25	< 0.001
Velvet Swimming Crab	2013<2017	Wind farm	0.23	< 0.001
Velvet Swimming Crab	2013<2017	Control	0.13	< 0.001

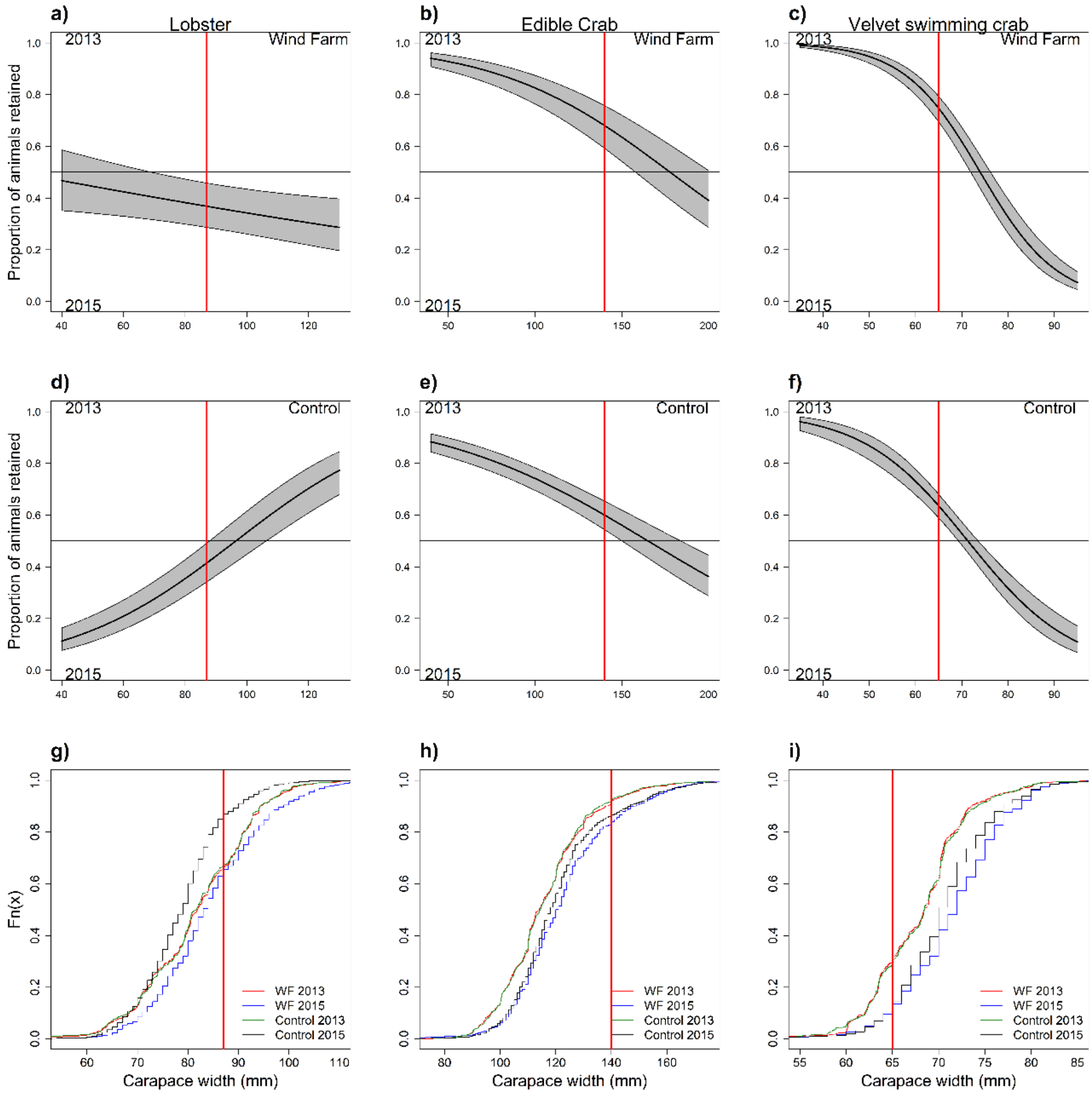


Figure 3.1: Plots derived from GLMM modelling of commercial species sampled at the WMR OWF site. a) Proportion of lobster, b) edible crab and c) velvet swimming crab in the wind farm site between **2013** & **2015**. d) Proportion of lobster, d) edible crab and e) velvet swimming crab in the **control** site between 2013 & 2015. The top box represents the baseline year (2013) and the bottom box represents Year 1 (2015). The grey shaded area represents the 95% confidence intervals and the bold black line the mean value. The central horizontal line represents the 0.5 (50%) value, points overlapping this line within the 95% confidence interval indicates that there was no significant difference in the proportion of an animal of that size between the two years. A value of 0.75 indicates that 75% of that species sampled at that size were sampled in 2013 and the other 25% sampled in 2015. The vertical red line represents the minimum landing size of the species within the fishery. This applies to all subsequent plots derived from GLMM modelling reported. g) Empirical cumulative distribution function (ECDF) plot of the lobster, h) edible crab and i) velvet swimming crab distribution for the **wind farm** and **control** site in 2013 (red and blue) and 2015 (green and black).The vertical red line on all plots represents the MLS of the species.

3.4.2.2 Edible crabs

The size frequency distributions of edible crabs sampled in the wind farm differed significantly between 2013 and 2015 (K-S, $p < 0.001$, Table 3.1), this was also the same for the control site (K-S, $p < 0.001$, Table 3.1). In 2013 there was a greater proportion of edible crabs up to 150 mm CW observed both in the wind farm (Figure 3.1b) and control sites (Figure 3.1e) than in 2015. Edible crabs between 150 - 180 mm CW did not show a difference in proportion between the two sample years, there was a slightly greater proportion of edible crabs > 180 mm CW observed in the control site in 2015 than in 2013. The ECDF plot (Figure 3.1h) demonstrated that within 2013 there was no observable difference in size distribution between the wind farm (red) and control (grey). There was an increase in size distribution (~130 – 150 mm CW) observed in 2015 in the wind farm (blue) than the control site (black).

3.4.2.3 Velvet swimming crabs

The size frequency distributions of velvet swimming crabs sampled in the wind farm differed significantly between 2013 and 2015 (K-S, $p < 0.001$, Table 3.1), this was also the same for the control site (K-S, $p < 0.001$, Table 3.1). Velvet swimming crabs up to ~ 70 mm CW were observed in greater proportion in 2013 across both sites than in 2015. Larger velvet swimming crabs (> 70 mm CW) were observed in greater proportion in 2015 at both the wind farm and control sites (Figure 3.1c & f). The ECDF plot (Figure 3.1i) demonstrates that there was no observable difference in the size distribution of velvet swimming crabs observed in 2013 across both sites (red and grey). Velvet swimming crabs demonstrated a greater size distribution in 2015, the wind farm site (blue) demonstrating a slightly greater distribution than the control site (black). This difference was observable between the MLS of 65 mm CW up to 80 mm CW. This is supported by the GLMM plots (Figure 3.1c & f) which demonstrates a greater proportion of larger velvet swimming crabs in 2015 across both sites.

Whilst size distribution of edible and velvet swimming crabs differed between the pre-construction survey and the first-year post build survey, the distribution in the wind farm between the two years followed the same trend as the control site. Indicating that factors other than the presence of the wind farm were responsible for the variation. Of the three species, lobsters showed the greatest response to the presence of the wind farm between the pre-construction survey and first-year post build surveys.

3.4.3 2013/2017 comparison

3.4.3.1 Lobsters

For all sites there was a significant difference in the size distribution of lobsters between 2013 and 2017 (K-S, $p < 0.001$; Table 3.1). This is not supported by the plot derived from GLMM in the wind farm. Figure 3.2a suggests that there was no difference in the size distribution of lobster sample in the wind farm between 2013 and 2017. For the control site the plot derived from GLMM (Figure 3.2d) suggests there was a greater proportion of lobsters less than ~75 mm CL sampled in 2017 than in 2013. The ECDF plots (Figure 3.2g) show that in the wind farm site there was a slightly greater size distribution of lobsters in 2017 up to approximately 80 mm CL. Above 80 mm CL the distribution shifts to their being a greater distribution of larger lobsters observed in 2013. The control site followed the same trend as the wind farm across 2013 and 2015.

3.4.3.2 Edible Crab

For all sites there was a significant difference in the size distribution of edible crabs between 2013 and 2017 (K-S, $p < 0.001$; Table 3.1). This was supported by the GLMM analysis. There was a greater proportion of edible crabs sampled in 2013 than in 2017 across all size ranges up to ~175 mm CW. Greater than 175 mm CW there was no difference in proportion between the two years (Figure 3.2b & e). The ECDF plots (Figure 3.2h) show that for the wind farm and control sites there was a slightly greater size distribution observed across the size spectrum in 2017 (blue and black) than in 2013 (red and grey). Indicating that although there was a greater proportion of edible crabs observed in 2013, they were of smaller size classes.

3.4.3.3 Velvet swimming crab

For all sites there was a significant difference in the size distribution of velvet crabs between 2013 and 2017 (K-S, $p < 0.001$; Table 3.1). At the wind farm site, there was a greater proportion of smaller velvet swimming crabs sampled in 2013, whereas in 2017 there was a greater proportion of larger velvet swimming crabs observed. The shift occurred just below the MLS of 65 mm CW. In the control site there was a greater proportion of velvet swimming crabs observed in 2017 across all size classes > 40 mm CW (Figure 3.2c & f). The ECDF plot (Figure 3.2i) shows that the wind farm in 2017 (blue) demonstrated the greatest difference in size distribution compared to 2013 (red) and also when compared to the control site in 2013 (green) and 2017 (black).

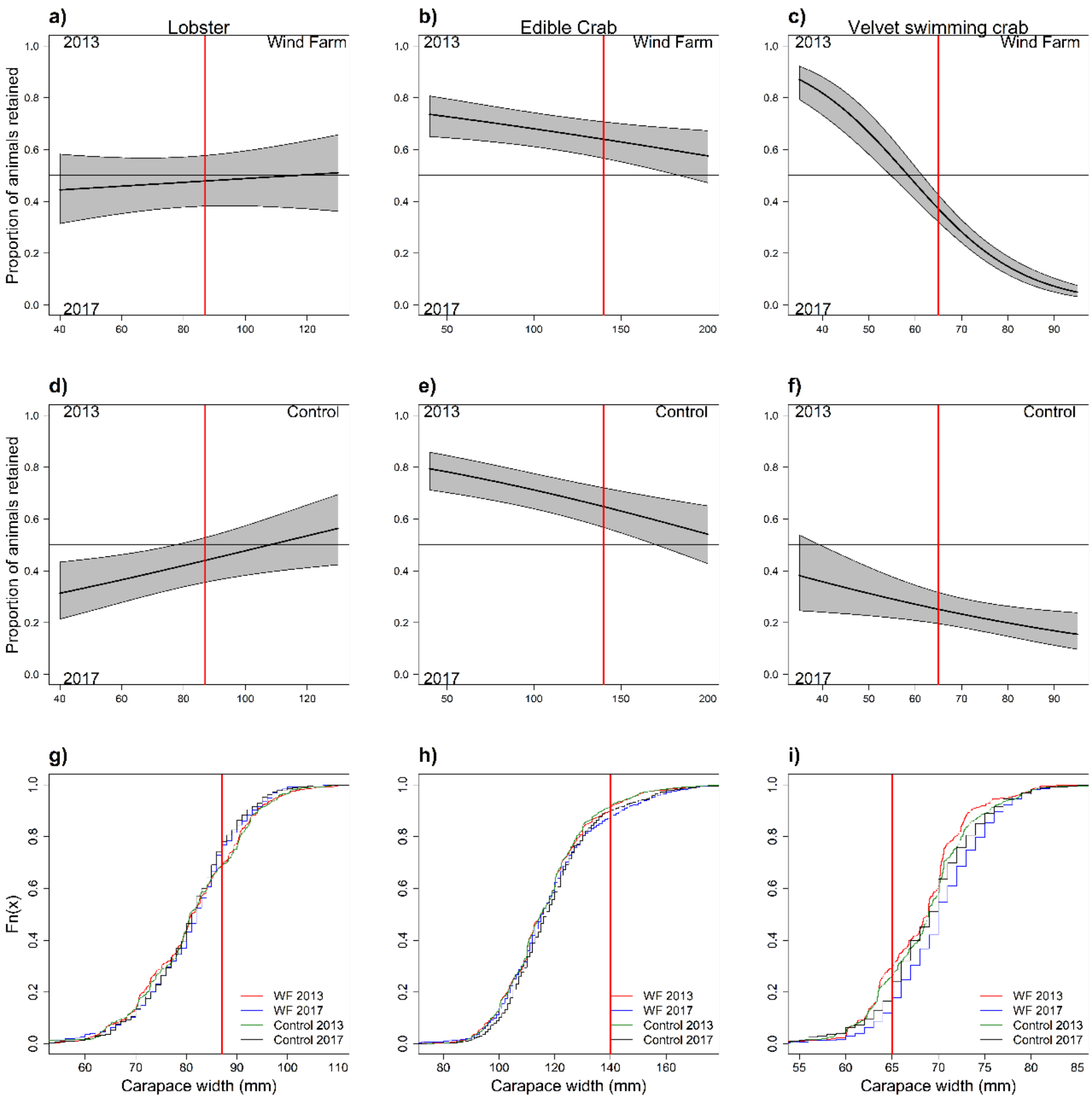


Figure 3.2: Plots derived from GLMM modelling of commercial species sampled at the WMR OWF site. a) Proportion of lobster, b) edible crab and c) velvet swimming crab in the wind farm site between 2013 & 2017. d) Proportion of lobster, e) edible crab and f) velvet swimming crab in the **control** site between 2013 & 2017. g) Empirical cumulative distribution function (ECDF) plot of the lobster, h) edible crab and i) velvet swimming crab distribution for the **wind farm** and **control** site in 2013 (red and blue) and 2017 (green and black). The vertical red line for edible crab (b, e & h) represents the new MLS of 140 mm CW (increased from 130 mm CW in Jan. 2016).

Between the baseline and third-year post builds surveys, lobsters and edible crab showed little response to the presence of the wind farm. Velvet swimming crabs showed the greatest response, demonstrating a greater proportion of larger velvet swimming crabs observed at both sites in 2017 compared to 2013 (Figure 3.2 c & f). However, this should be taken in context with the change in fisheries management, introducing mandatory escape gaps in all commercial pots in the area, allowing escape of velvet swimming crabs.

Therefore, the null hypotheses ‘‘Size structure of lobsters and associated commercial by-catch did not differ significantly between the OWF and control over the three survey years’ was rejected.3.4.2 Catch and landings per unit of effort

3.4.3.4 Lobsters

There was no significant difference in mean CPUE of lobsters (between the wind farm and control site and no significant interaction between Site and Year (ANOVA, $p > 0.05$; Table 3.2). However, there was a significant difference in mean CPUE of lobsters between years (ANOVA, $p < 0.001$; Table 3.2). There was a significantly greater mean CPUE in 2015 than in either 2013 or 2017 (Tukey, $p < 0.05$, Figure 3.3a, Table 3.2). Mean LPUE of lobsters also differed significantly between years and between sites (ANOVA, $p < 0.001$: Table 3.4,Figure 3.3b). Mean LPUE of lobsters was significantly greater in 2015 than 2017 and also mean LPUE of lobsters in the wind farm in 2017 was significantly greater than all other cases (Tukey, $p < 0.05$). There was also a significant interaction between site and year (ANOVA, $p < 0.001$: Table 3.2). This was predominantly due to a greater mean LPUE in the Wind farm site in 2015 (Tukey, $p < 0.05$, Figure 3.3b). This was highlighted by a greater ratio of LPUE/CPUE in the Wind farm site in 2015 (Figure 3.3b).

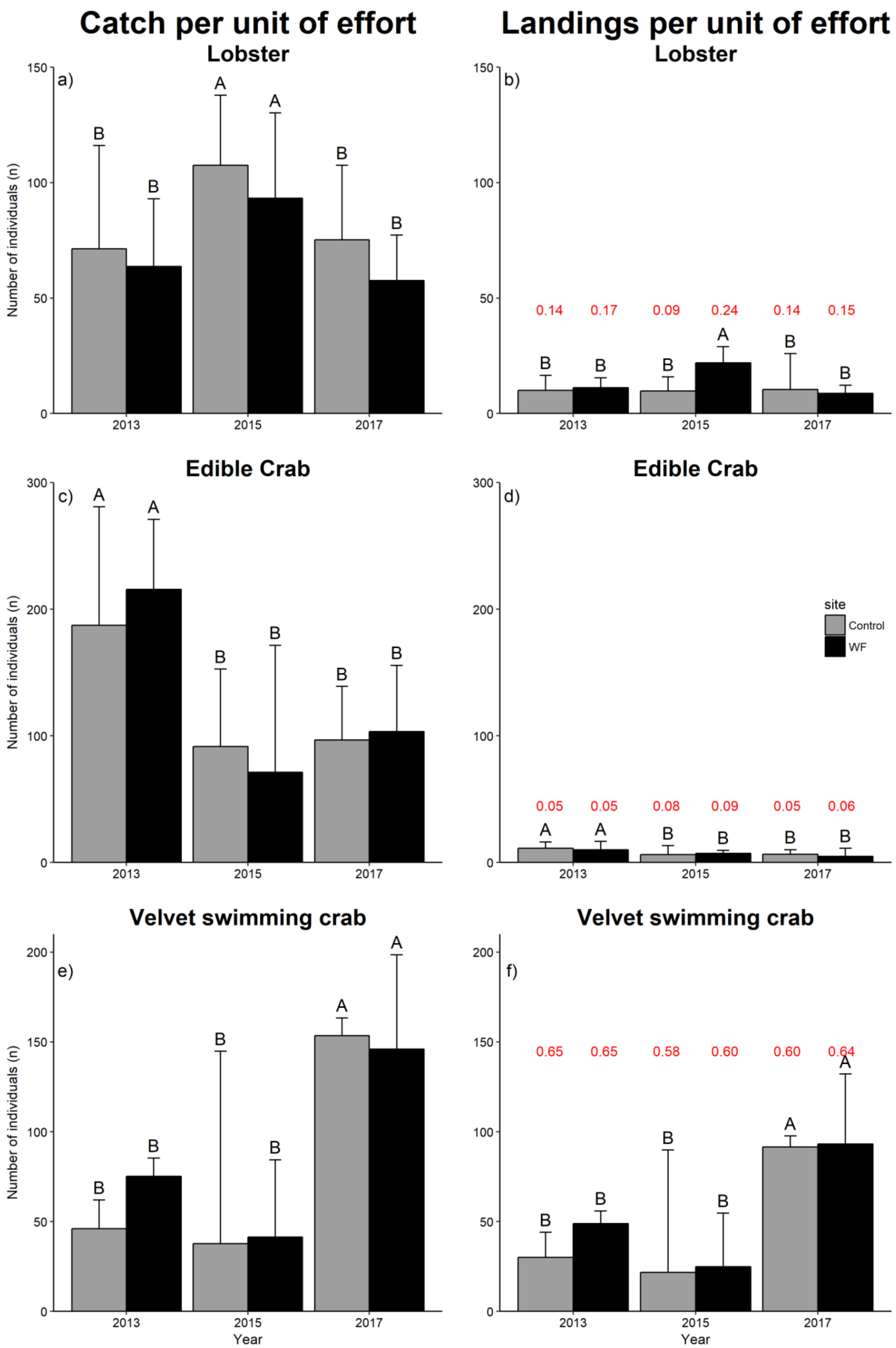


Figure 3.3: Bar plots of catch per unit of effort for lobster and the number of landable individual per unit of effort for lobster a) & b), edible crab c) & d) and velvet swimming crab e) & f). Data are mean plus standard deviation. Red numbers on right-hand plots are the ratio between the CPUE and LPUE for each corresponding site and year.

3.4.3.5 Edible Crabs

There was no significant difference in mean CPUE of edible crabs between either the wind farm and the control and no significant interaction between site and year (ANOVA, $p > 0.05$; Table 3.2). Mean CPUE of edible crabs differed significantly between years (ANOVA, $p < 0.001$; Table 3.2), there was a significantly greater mean CPUE of edible crabs in both the wind farm and control sites in 2013 than in 2015 or 2017 (Tukey, $p < 0.05$, Figure 3.3c;). This was not evident in the mean LPUE of edible crabs. There was no significant difference in mean LPUE of edible crabs between sites or year. There was also no significant interaction between site and year (ANOVA, $p > 0.05$: Figure 3.3d, Table 3.2).

3.4.3.6 Velvet crabs

There was no significant difference in mean velvet crab CPUE between the wind farm and control site (ANOVA, $p > 0.05$; Table 3.2). Mean velvet swimming crab CPUE was significantly greater in 2017 than in either 2013 or 2015, (ANOVA, $p < 0.01$; Table 3.2). There was a significant interaction between site and year (ANOVA, $p < 0.01$; Table 3.2), mean velvet swimming crab CPUE was significantly greater at both the wind farm and control site in 2017 than all other cases (Tukey, $p < 0.05$) (Figure 3.3e, Table 3.2). Mean LPUE of velvet crabs did not differ significantly between years (ANOVA, $p > 0.05$; Table 3.2). Mean LPUE of velvet swimming crabs was significantly greater at both the wind farm and control sites in 2017 than in either 2013 or 2015 (Tukey, $p < 0.05$). There was a significant interaction ($p < 0.05$; Table 3.2) between site and year and a significant difference of mean LPUE between years (Figure 3.3f). Although mean LPUE in the wind farm sites differed significantly between years, the ratio between corresponding CPUE/LPUE showed little variation (0.06) between years suggesting that changes in size distribution were not impacting on catches.

Year was the dominant factor when analysing CPUE and LPUE, this was the case for both lobsters and velvet crabs. Therefore, the null hypothesis '*catch rates of lobsters and associated commercial by-catch did not differ significantly between the OWF and control over the three survey years'* was rejected.

Table 3.2: Results of two-way repeated measures ANOVA testing CPUE and LPUE between site (wind farm and control) and year (2013/15/17) for all three-commercial species sampled during the WMR survey. Results of post-hoc analysis (Tukey test) presented.

Species	Effort	Factor	F Value	*p*	Differences (Tukey test)
Lobster	CPUE	Site	0.161	n.s.	n.s.
		Year	**15.187**	**< 0.001**	**2015 > 2013 2015 > 2017**
		Site*Year	0.260	n.s.	n.s.
Lobster	LPUE	**Site**	**16.31**	**< 0.001**	**2015 WF > all other sites**
		Year	**33.03**	**< 0.001**	**2015 > 2017**
		Site*Year	**24.07**	**< 0.001**	**2015 WF > 2015 C 2015 WF > 2013 & 2017 both sites**
Edible Crab	CPUE	Site	2.019	n.s.	n.s.
		Year	**35.703**	**< 0.001**	**2013 > 2015 2013 > 2017**
		Site*Year	2.697	n.s.	n.s.
Edible Crab	LPUE	Site	0.480	n.s.	n.s.
		Year	0.059	n.s.	**2013 > 2015 2013 > 2017**
		Site*Year	0.001	n.s.	n.s.
Velvet Crab	CPUE	Site	0.407	n.s.	n.s.
		Year	**44.564**	**< 0.001**	**2017 > 2013 2017 > 2015**
		Site*Year	**4.502**	**< 0.05**	**2017 WF & C > both sites in 2013 & 2015**
Velvet Crab	LPUE	Site	3.271	n.s.	n.s.
		Year	**82.708**	**< 0.0001**	**2017 > 2013 2017 > 2015**
		Site*Year	**3.085**	**< 0.05**	**2017 WF & C > both sites in 2013 & 2015**

3.4.4 Bycatch Diversity

There was no significant difference in median species richness between years in either the wind farm or the control sites (Kruskal Wallis, p < 0.05, Figure 3.4a, Table 3.3). There was also no significant difference in median species richness between the wind farm and control within each individual survey year (Wilcoxon rank sum test, p > 0.05, Figure 3.4a Table 3.3).

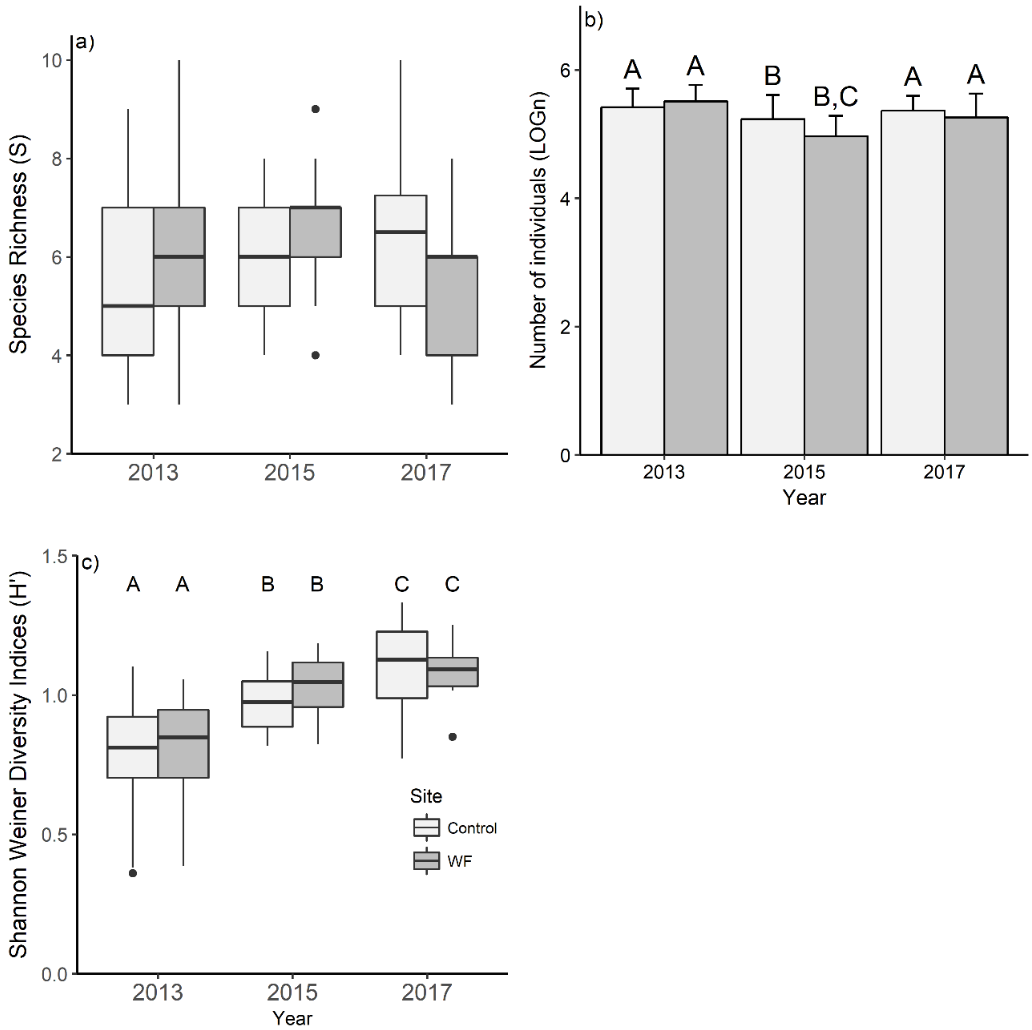


Figure 3.4: a) Species richness for both the control and wind farm sites for all three survey years. The points represent the individual data points recorded, e.g. the median daily species richness recorded. b) The mean (Log) number of individual (species abundance) recorded at the sample sites/years. c) The median Shannon Wiener diversity index of the sample sites/years. *For the b) the top of the bar represents the mean and the error bars represent the standard deviation around the mean. For the boxplots (a & c) presented the median of the value of y is represented by the bold horizontal line; the top of the box above the median represents the 75th percentile with the bottom of the box representing the 25th percentile. The whole box represents the interquartile range (25-75), the whiskers represent the maximum and minimum values of y. The outliers represent data points that were greater than 1.5 times the interquartile range above the third quartile or below the first quartile. This applies to subsequent box plots reported in this study.*

Table 3.3: Results of non-parametric tests on species richness (S) and Shannon Wiener diversity indices (H') between survey years and also between the wind farm and control sites.

Variable	Treatment	Test	Test statistic	p value	Differences (Pairwise wilcox test)
Species richness (S)	WF between years	Kruskal Wallis	4.52	n.s.	n.s.
	Control between years	Kruskal Wallis	3.39	n.s.	n.s.
	2013 between WF and Control	Wilcoxon rank sum	234	n.s.	n.s.
	2015 between WF and Control	Wilcoxon rank sum	226	n.s.	n.s.
	2015 between WF and Control	Wilcoxon rank sum	173.5	n.s.	n.s.
Shannon Wiener diversity indices (H')	**WF between years**	**Kruskal Wallis**	**28.14**	**< 0.001**	**2013 < 2015 2013 < 2017 2015 < 2017**
	Control between years	**Kruskal Wallis**	**25.98**	**< 0.001**	**2013 < 2015 2013 < 2017 2015 < 2017**
	2013 between WF and Control	Wilcoxon rank sum	246	n.s.	n.s.
	2015 between WF and Control	Wilcoxon rank sum	179	n.s.	n.s.
	2015 between WF and Control	Wilcoxon rank sum	144	n.s.	n.s.

LOGn was significantly greater in the control site than the wind farm site during 2015 2017 (ANOVA, $P < 0.01$, Figure 3.4b, Table 3.4). LOGn was not significantly different between the wind farm and control sites in either 2013 or 2017 (ANOVA, $p > 0.05$). LOGn was significantly less in 2015 than in either 2013 or 2017 (ANOVA, $P < 0.01$, Figure 3.4b, Table 3.4). There was also a significant interaction between site and year, LOGn in the wind farm in 2015, was significantly less than all other cases (Tukey, $p < 0.05$, Table 3.4)

Median Shannon Weiner diversity indices differed significantly between years for both the wind farm and control site (Kruskal Wallis, $p < 0.001$, Figure 3.4c, Table 3.3). Median Shannon Weiner diversity indices increased significantly between 2013 and 2015 and again between 2015 and 2017 in both the wind farm and control sites (Figure 3.4c, Table 3.3). Median Shannon Weiner diversity indices did not differ significantly between the wind farm and control in any of the individual survey years (Wilcoxon Rank Sum, $p > 0.05$, Figure 3.4c, Table 3.3).

Table 3.4: Results of two-way repeated measures ANOVA testing the total abundance (LOGn) of bycatch individuals sampled between all three survey years and between the wind farm (WF) and control (C) sites.

Variable	Treatment	Factor	F Value	p	Differences (Tukey test)
Number of individuals (LOGn)	Site and Year	Site	5.83	< 0.05	C 2015 > WF 2015
		Year	16.17	< 0.001	2013 > 2015 2017 > 2015
		Site*Year	4.81	< 0.001	WF 2013 > C 2015 WF 2015 < all other cases

The null hypothesis '*Species richness did not differ significantly between the wind farm and control site over the 3 survey years*' was accepted, however, the null hypothesis ' Species *abundance, diversity and community assemblage did not differ significantly between the wind farm and control site over the 3 survey years*' was rejected.

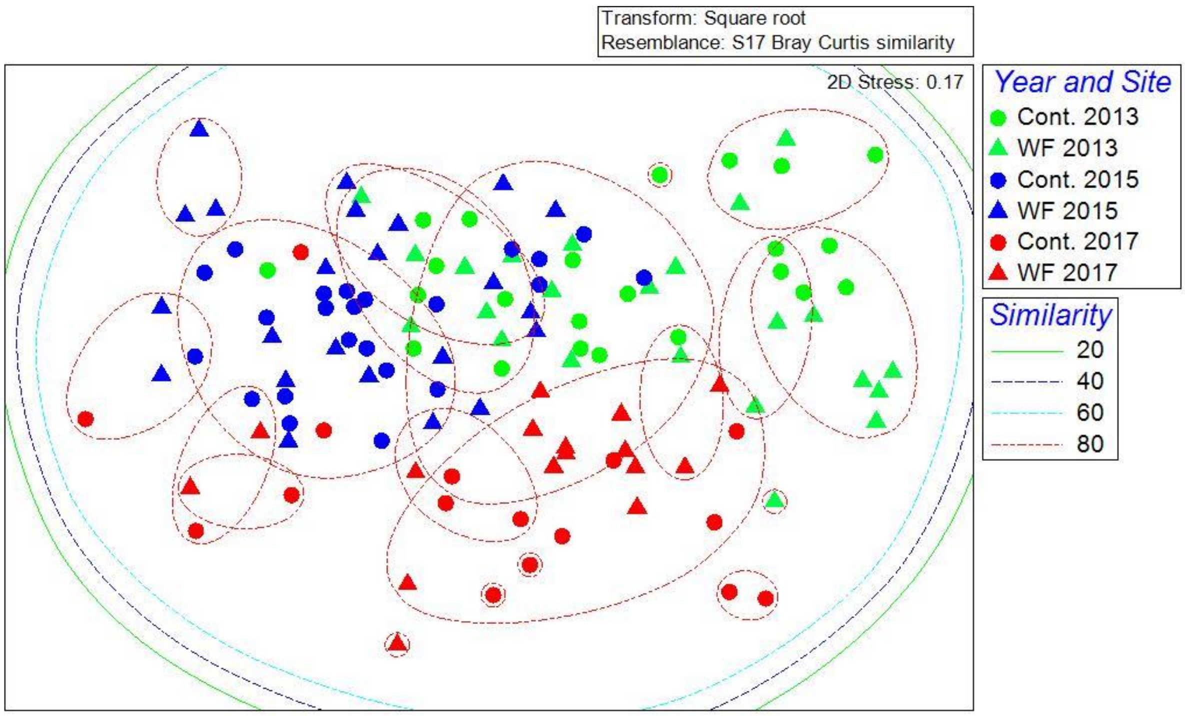


Figure 3.5: The results of MDS analysis on a Bray-Curtis similarity matrix of the bycatch communities observed in lobster pots sampled at the WMR OWF site. The Year and Site data represents each sample year and both the Wind Farm and the Control Sites within each year. There were three distinct clustering of points representing each of the sample years with slight overlap between 2013 and 2015. Within each year there was no distinct clustering of points between the wind farm and control sites. The stress value of the 2D plot is 0.17 indicating that this was a reasonable representation of the similarity between the three sample years and sites. Points are grouped using results of similarity clustering.

There was a significant difference in bycatch community assemblages observed at the WMR OWF between the three sample years (ANOSIM, Global R = 0.321, p = 0.1%, Figure 3.5). There was no significant difference in bycatch communities between the wind farm and control in 2013 and 2017, in all other cases the bycatch communities differed significantly (Table 3.5).

Table 3.5: Results of ANOSIM on Bray-Curtis similarity matrix of the communities sampled in pots over all three years and the wind farm and control sites, at the WMR OWF. Significant results are **bold**.

Group statistics	Global R	p (%)
2013 WF / Control	0.002	35.6
2015 WF / Control	**0.065**	**2.5**
2017 WF / Control	0.028	15.5
2013 WF / 2015 WF	**0.423**	**0.1**
2013 WF / 2017 WF	**0.251**	**0.1**
2015 WF / 2017 WF	**0.433**	**0.1**
2013 Control / 2015 Control	**0.351**	**0.1**
2013 Control / 2017 Control	**0.408**	**0.1**
2015 Control / 2017 Control	**0.483**	**0.1**

Non-commercial bycatch (below their respective MLS or > MLS but of poor quality) of lobsters, edible crab and velvet swimming crab were the dominant species accounting for similarity in community assemblages between years (Table 3.6). Non-commercial edible crab bycatch had the greatest average abundance across all three years and contributed the greatest to the dissimilarity between sites (Table 3.6).

Table 3.6: Results of SIMPER analysis, highlighting dominant species contributing to the similarity between sites (90% cut off for low contributors).

	Lobster	Edible crab	Velvet swimming crab
Average abundance 2013	51.75	171.76	19.59
Average abundance 2015	81.80	68.17	15.09
Average abundance 2017	59.78	94.16	57.54
% contribution to dissimilarity 2013/2015	25.38	63.62	6.16
% contribution to dissimilarity 2013/2017	19.17	53.31	22.79
% contribution to dissimilarity 2015/2017	25.65	39.25	28.99

3.5 Discussion

3.5.1 Size distribution

Lobster size distribution changed significantly between the three sample years. The most significant change was between 2013 and 2015. In the wind farm there was a greater proportion of larger lobsters sampled in 2015 than in 2013 whereas the opposite was observed in the control site. The closure of the wind farm to fishing exploitation meant that lobsters above the MLS were not removed from the areas due to fishing. This was not observed when the same sites were compared between 2013 and 2017. The plot derived from GLMM analysis demonstrates that there was no difference in the proportion of lobsters sampled in the wind farm between 2013/17. However, the K-S test and ECDF plot comparing 2013 and 2017 length frequency distributions did not support this. The disparity between the two analyses/plots should take into account that due to its limitations (assuming equal hauls) the GLMM used sub-sampled data (2013) whereas the KS and ECDF represent the whole data set for 2013/17. At the control site there was slightly greater proportion of smaller lobsters observed in 2017 (< 75mm CL). The change in size structure of the lobsters sampled between the two sites, over the three sample years can be attributed to several factors. Closure of the wind farm site in 2015 due to the construction of the wind farm excluded fishers from the area for a period of 20 months. Fishers could still fish in the control area for this period. Closure of areas to fishing exploitation has been demonstrated to have a positive effect on the biomass of lobster populations (Goñi *et al.*, 2010). For example, the Lundy Island NTZ reported an increase in abundance and size of lobsters in the area post designation of the site (Hoskin *et al.*, 2011). The effect of closed areas on crustacean stocks is the focus of Chapter 4.

Lobster dominance and competition for resources is generally governed by overall body size (Phillips, 2007). The survey sites will maintain a limited carrying capacity for crustacean populations and are affected by factors such as, but not limited to: nutrient availability, habitat complexity, shelter, competition, mate availability and fishing pressure (Caddy & Stamatopoulos, 1990). Within the wind farm, prior to construction, boulders were removed from the site. This could have reduced the availability of shelters in the area. Thus, increasing the competition for shelter and potentially displacing smaller lobsters from the area. Additionally; size of shelters has been demonstrated to be a limiting factor governing growth rates in lobsters (Émond *et al.*, 2010) and lobster density can be affected by shelter availability (Ball *et al.*, 2001; Steneck, 2006). These effects are unlikely due to the addition of scour stone protection which may offset this habitat loss (Wilson and Elliott, 2009), and has been shown to be a suitable habitat for macrobenthic crustaceans (Krone *et al.*, 2017) The greater proportion of larger lobsters in the wind farm, due to the lack of fishing pressure, could have influenced the catch dynamics during the sampling. Pots can be used as temporary habitats for smaller lobsters, providing shelter, refuge from predation and nutrients in the form of bait and biofouling (Smolowitz, 1978). If a larger lobster encounters the pot first and is subsequently caught, it can deter other smaller crustaceans from entering the pot. These pot dynamics may have skewed the data to under-represent the abundance of smaller lobsters present within the wind farm. Additionally size of lobster territory/home range has been loosely linked to their size (Smith *et al.*, 2001). The presence of larger lobsters at the site, with a greater home range, due to the absence of fishing pressure, may have deterred smaller lobsters from the area, altering the size structure of the wind-farm population. The absence of fishing pressure in 2015 would only directly affect lobsters above MLS as observed in the OWF during this period as lobsters below MLS would have been discarded back on site during routine fishing practices. The control site in both 2015 and 2017 demonstrated a greater proportion of smaller lobsters when compared to 2013. This indicates that there was potential overspill/displacement of smaller lobsters into the control site due to the control site not being subjected to factors affected by lack of fishing exploitation. This supports the findings of Hoskin *et al.* (2011) who observed limited overspill of smaller lobsters from the Lundy Island NTZ into adjacent areas.

Edible crab size distribution followed the same trend over both sites and all three sample years. There was generally a greater proportion of edible crab sampled in 2013 than either 2015 or 2017. The proportion of larger edible crabs (> ~160 mm CW), showed no difference between years in each site. The ECDF plots demonstrate that although there were significantly more edible crabs sampled in 2013, they were of smaller size classes than observed in either 2015 or 2017. Due to the area being a mixed fishery, as with lobsters, there are several variables that affect an edible crab entering a pot (Jury and Watson, 2013). Primary amongst these is the presence of lobsters within a pot prior to an edible crab encountering it. The lower proportion of edible crabs observed in 2015/17 may have been affected by the increased presence of lobsters at the wind farm and control site during 2015 and 2017, which resulted in the higher CPUE reported across both sites for 2015/17 than in 2013. However, as the control sites reflected the same pattern as the wind farm sites, it can be assumed that this can be attributed to natural variability within the fishery.

The size distribution of velvet swimming crabs was smaller across both sites in 2013 than in either 2015/17. There was greater proportion of larger velvet swimming crabs observed in both 2015 &17 than in 2013. The difference in proportion of velvet swimming crabs between years was generally split around the MLS (65 mm CW) with the exception of the control site in 2017 – a greater proportion observed in 2017 of velvet swimming crabs > 40 mm CW. The difference between 2013 and 2015 could be attributed to the greater abundance of edible crabs observed in 2013. The greater abundance of smaller size classed edible crabs could have occupied a similar niche as the velvet swimming crabs and outcompeted them for resources. Velvet swimming crabs will also be affected by the pot dynamics previously discussed. The difference between 2013 and 2017 can be attributed to a management measure being introduced in January 2016. This measure ensured that all pots within 6 nm of the baseline were to be fitted with two escape gaps measuring 80/46 mm in dimensions (NEIFCA 2015). This introduction, whilst introduced to reduce bycatch, inadvertently allowed "landable" velvet swimming crabs to escape commercial pots. The survey sites were located within the jurisdiction of this byelaw and all commercial pots in the area had to fit escape gaps, thus seeing a reduction in their velvet swimming crab catch (Ackers, per. comm. Manager, ISC). The pots used in this survey were exempt from this measure, therefore it is likely that the increased escape of velvet swimming crabs from commercial pots affected/increased the catch rates of the pots used within the survey. The increase in velvet swimming crab catch reported is likely as a result of the introduction of escape gaps.

3.5.2 Catch statistics

Catch per unit of effort (CPUE) for all three-species varied significantly between the three sample years, although there was no significant difference between windfarm and control sites within each year. Edible crab and velvet swimming crab showed a significant interaction between year and site however lobster showed no significant interaction between year and site. Lobster CPUE was greatest across both sites in 2015, edible crab CPUE was greatest in 2013 and velvet swimming crab CPUE was greatest in 2017. CPUE of edible and velvet swimming crab was lowest in 2015 whilst lobster was at its greatest. This further supports the theory of lobsters affecting catch rates of other commercial shellfish species in a mixed fishery. The lack of fishing pressure in the wind farm allows for a significantly greater CPUE of lobsters in 2015 in comparison to other years. However, the control site demonstrated a greater CPUE of lobsters than the wind farm in 2015. This could be evidence of overspill/displacement from the wind farm during the closure or just natural variability. The number of landable lobsters (LPUE) in 2015 was significantly greater in the wind farm than in the control site. This means that the greater abundance of larger, good quality lobsters could be affecting the smaller, poor quality or "soft" lobsters with regards to pot dynamics or displacement from the site. The CPUE and LPUE of both sites between 2013 and 2017 showed little variation. The variation in 2015 can be attributed to the wind farm site being closed to fishing exploitation.

Edible crab CPUE was greatest in 2013 and lowest in 2015 at both the wind farm and control site. Although the CPUE of edible crabs was at its lowest in the wind farm site in 2015, the ratio between LPUE/CPUE in the wind farm was at its greatest (0.03 greater than other years). This is a relatively small increase in LPUE, although it indicates fishing exclusion within the wind farm increasing the LPUE of edible crabs. The significantly greater CPUE observed in 2013 did not translate into a greater LPUE in 2013, there was no significant difference in LPUE between years and little variation in the LPUE/CPUE ratio. This indicates that the higher CPUE observed in 2013 was made up of edible crabs that were not suitable for landing. These were edible crabs of smaller size classes and poorer quality (determined by conditioning index presented in Table 2.1) thus resulting in a low LPUE. No significant difference of LPUE between years of edible crabs indicates that the development of the wind farm has had no observable effect on the edible crab fishery during the sampling season. LPUE was generated accounting for the change in MLS from 130 -140 mm CW introduced in Jan. 16. Therefore, there may have been slightly greater LPUE in 2017 in the size class 130 – 140 mm CW, however as there was no discernible difference is size distribution reported it can be assumed that the LPUE comparison reflects the population structure of edible crabs.

Velvet swimming crab CPUE and LPUE were significantly greater in 2017 in comparison to 2013/15. These observable increases could be due to the velvet swimming crabs taking advantage of the additional habitat created by the scour stone protection around the monopile bases. It has been demonstrated that scour stone protection can support extensive crab populations (Krone *et al.*, 2017). However, as there is no significant difference between the wind farm and control sites it is likely that the introduction of escape gap (which allowed increased escape from commercial pots, resulting in greater catch rates in the survey pots that were exempt from the byelaw) was likely responsible for this increase.

Both the size and CPUE/LPUE results for edible and velvet swimming crab should be taken into context with the survey design. This survey was designed to target the commercially important lobster fishery, both in bait used and time of year sampled. These results demonstrate the commercial bycatch of edible and velvet swimming crabs in the region.

Whilst this chapter has highlighted differences in size structure of lobsters and commercial bycatch at the Westermost Rough OWF site and associated control, leading to an acceptance of Hypothesis 3.1, this acceptance of the hypothesis should consider the variables discussed above leading to these effects.

3.5.3 Diversity

Cluster analysis of the bycatch demonstrated three clusters representing each year with slight overlap between the three years. Within each year, there was overlap between the wind farm and control sites, however the similarity clustering does highlights that all sites and years fall within 60 % similarity of each other. Subsequent analysis of similarities (ANOSIM) highlighted that there was a significant difference in bycatch community composition in all cases, except for between the wind farm and control in both 2013 and 2017. This indicates that the reduction in fishing pressure in 2015 led to a change in community composition due to an increase in overall size, CPUE and LPUE of lobsters in the wind farm and a reduction in the number of individuals sampled. ANOSIM also supports the increase in diversity reported. However, as the Shannon Weiner diversity indices saw an increase between 2013, 2015 and 2017 at both the wind farm and control site it is likely to represent a natural variation in species assemblage or increase in biodiversity at the sites as opposed to the effects of wind farm construction. Therefore Hypothesis 3.2 can be accepted, there was an observable affect to the non-commercial bycatch community assemblage between the wind farm and control site over the three survey years.

Short term disturbance has been found to have a positive effect on some marine habitats (Thrush *et al.*, 1995). For example, edible crab abundance has been shown to increase significantly over a short period in areas immediately following bottom dredging (Ramsay *et al.* 1998; Jenkins *et al.* 2001). This may explain the increase in biodiversity observed in this survey. However, the species that are likely to be recorded using the reported sampling regime are only those likely to be captured using a pot. Additionally, the fish species that may be attracted to the fish aggregation device properties of an OWF may not necessarily get caught in a lobster pot. Therefore, it is unlikely that the design of this survey will accurately reflect whether the installation and operation of the WMR OWF has had a positive or negative effect on biodiversity in the area.

3.6 Conclusions

This study has highlighted that the building of an OWF has short-term effects (within 3 years) on the ecology of the lobster population and the commercial and non-commercial bycatch in the area. Effecting the size and catch rates of lobsters and associated commercial bycatch. These changes could be attributed to the construction and subsequent operation of the wind farm however it is more likely that the influence of the exclusion of fishing effort during the construction phase was the dominant factor. The key result was the increased proportion of smaller lobsters observed in the control site in 2015/17 and their lower proportion in the wind farm over this period. This indicates a shift towards smaller lobsters within the wind farm site. The differences observed between the baseline survey in 2013 and 2015 were not as defined in 2017 indicating that over the short period between surveys the wind farm site was starting to reflect the control site and surrounding area. This demonstrates that although there were observable effects in the size, catch rates and diversity pre, and post-construction of the wind farm, the site is starting to reflect the surrounding area.

Chapter 4: A study on the effects of exclusion of fishing effort due to offshore wind farm construction on a commercially important crustacean fishery.

4.1 Abstract

Closed areas are often a tool used for fisheries management for conservation of commercially important stocks. There are detrimental factors to permanently closed areas such as displacement of effort and increased inter and intra-species competition. The closure of the Westermost rough offshore wind farm site in the north-east of England during construction allowed an investigation into the effects of the closure on the ecology of commercial shellfish populations in the area and the effects of subsequently reopening of the site.. Closure of the site saw an increase in size and catch per unit of effort of legal-sized *Homarus gammarus* but potential displacement of smaller *H. gammarus* and *Cancer pagurus*. Reopening of the site saw a rapid, short term increase in fishing exploitation, leading to a decreased CPUE and shift in size structure in the offshore wind farm site, this did not reach levels below that of the control site. The temporary/rotational closure of selected areas can be ecologically beneficial and offer a management option for crustacean fisheries.

4.2 Introduction

Closed areas are often used in marine spatial planning for the conservation of habitats or specific species. There has been a paradigm shift to the concept of marine protected areas (MPAs) as one of the most effective tools for marine conservation. The legislation involved with MPAs varies, depends on the aims of the designated site and as such can be misleading. For example, Nicoll and Day (2017) argued that the Commission for Antarctic Marine Living Resources (CCAMLR) convention area, whilst thought to be equivalent to a class IV MPA (habitat and species management area) (IUCN, 2008) does not meet this requirement as the convention focuses primarily on fisheries and not the wider conservation of nature . The UK aims to attain a comprehensive network of marine conservation zones (MCZs) in accordance with goals set out in the Marine and Coastal Access Act, 2009. There are currently 91 MCZs designated within UK waters legislating 27,227 km^2 of seas (DEFRA, 2019a). The designation of MCZs in the UK was for varying reasons, for example, protection of intertidal and sub-tidal habitats, IUCN protected species and species and habitats susceptible to disturbance (DEFRA, 2019a).

Restrictions of specific fishing effort types are often one of the main goals of implementing an MPA. Mobile gear such as bottom trawling can be destructive to fragile habitats, scallop dredging has also been demonstrated to have a detrimental effect on marine flora and fauna (Thrush *et al.*, 1995; Jenkins *et al.*, 2001). However, short term disturbance due to bottom dredging can benefit certain species such as *Cancer pagurus* (Ramsay *et al.*, 1998). The Holderness Inshore MCZ was designated in 2016 to maintain in favourable condition the intertidal, subtidal and circalittoral rocks and sediments (DEFRA, 2019a). Mobile fishing gear is excluded from the area whereas static gear fisheries are permitted (e.g. pot and static net fisheries). However, there is no specific legislation within this MCZ excluding mobile gear types, this is legislated against under a local byelaw (NEIFCA, 2003). Legislation of UK MCZs can fall under the remit of regional management authorities such as NEIFCA and national bodies such as the MMO, Natural England, Environment Agency, Department for Trade, local harbour authorities and the Oil and Gas Authority, following guidance from JNCC (DEFRA, 2019a). They are tasked with implementing management measures that account for the MCZ designation and the needs of the local communities such as commercial fishing.

Protected areas can enhance localised marine biodiversity by the spill-over effect (Goñi *et al.* 2010; Huserbråten *et al.* 2013). This is a process by which the protected stocks can enhance surrounding areas via immigration into and emigration out of a protected site and provide increased production and recruitment from the site. Krone *et al.,* (2017) observed an increase of *C. pagurus* within an OWF (acting as a quasi no-take zone

(NTZ) due to absence of mobile fishing effort) which was predicted to enhance the recruitment into the population by 27%. The potential positive effects of spill-over can be difficult to ascertain due to the temporal scale of studies not being of sufficient length to accurately capture the effect (Moland *et al.*, 2013b; Smyth *et al.*, 2015; Vandendriessche *et al.*, 2015). Spill-over effects have been observed in different lobster populations within a closed area over a period of ten years for *Palinurus elephas* (Goni *et al.*, 2003) and four years for *H. gammarus* (Hoskin *et al.*, 2011). The spill-over effect can lead fishers to "fish the line", a process by which fishing intensity around a protected area is increased to take advantage of this process. Spatial displacement of effort, specifically in static fisheries where fishers can have strong fidelity to specific sites (Hart and Johnson, 2002b; Turner *et al.*, 2013), can increase pressure in surrounding areas.

No Take Zones are a more stringent form of protected area and can be more effective than other MPA types (Long, 2017). Their purpose is to prevent the removal of any natural resources from the area. Designation of NTZs is most efficient when they protect sessile species or species with small home ranges, for example lobsters, that demonstrate strong site fidelity (Bannister and Addison 1998; Smith *et al.* 1998; Moland *et al.* 2011), although these species can migrate offshore seasonally to deeper water during colder months (Caddy, 1986). Within UK waters there are only three designated NTZs, Lundy, Flamborough Head and Lamlash Bay (JNCC, 2011). Lundy Island was designated as a Marine Nature Reserve in 1986, an NTZ in 2003 in accordance with the Marine and Coastal Access Act 2009 and an MCZ in 2010. The purpose of the designation was to ensure the spiny lobster (*Palinurus elephas*) recovers to a favourable condition (DEFRA, 2019a). The Lundy Island NTZ is the oldest MCZ in the UK and has been the subject to a variety of studies looking at the effects of its implementation. The Lundy Island NTZ was observed to have an increase in European lobster (*Homarus gammarus*) abundance and biomass (Hoskin *et al.* 2011; Wootton *et al.* 2012; Davies *et al.* 2015) in the period since its designation. However further study of the site highlighted detrimental effects of the closure such as increased injury in larger lobsters, displacement of smaller lobsters, reduction in *Necora puber* abundance and increased disease in lobsters (Hoskin *et al.*, 2011; Wootton *et al.*, 2012; Davies *et al.*, 2015). The absence of fishing pressure has removed top-down control on the lobsters that were of legal size within the fishery. Lobsters are territorial and demonstrate strong inter-species competition, competing for resources such as shelter, nutrients and mates (Ball *et al.*, 2001; Steneck, 2006). Lobster dominance is based on size of animal (Wahle *et al.*, 2013), thus the greater abundance and biomass observed due to fishing exclusion in the area may have altered the dynamics within the Lundy Island NTZ ecosystem. This was supported by Howarth *et al.*, (2017) who reported an increase in catch and weight per unit of effort inside the Lamlash Bay NTZ when compared to the surrounding fished area.

Implementation of protected areas has often met with resistance from the commercial fishing industry (Kaiser, 2005). The ecological benefit of a protected area may result in an economic benefit to the fisher via mechanisms such as over-spill , this is often met with scepticism from commercial fishers (Leleu *et al.*, 2012; Caveen *et al.*, 2014). This is predominantly due to the implementation of surveys assessing the effects of closed areas not reflecting the fishing effort in the area and the data are rarely published (Hooper and Austen, 2014b; Hooper *et al.*, 2015). However, whilst there is extensive stakeholder engagement led by spatial managers such as online consultations and open forums, engagement from the fishing industry can be quite low and opposition to sites often lacks evidence to support it (DEFRA, 2019b). Closed areas can be a contentious issue in both the scientific and commercial fishing industries and are often perceived by members of both sectors to be implemented for political reasons as opposed to ecological or conservation purposes (Kaiser, 2005). However, they are often used successfully to manage marine resources and a popular tool for spatial and resource managers (Long, 2017; Sala and Giakoumi, 2018; Boudouresque *et al.*, 2019). The use of closed areas as a fisheries management tool should be treated as a rigorously designed experiment, conducting accurate cost/benefit analysis (Kaiser, 2005; Caveen *et al.*, 2014).

The concept of rotational harvest can offset potential issues with permanently closed areas and can assuage some of the concerns of commercial fisheries. Rotational harvest uses temporarily closed areas/fisheries to ease the fishing pressure on a selected stock for an optimum period. This area/fishery is then subsequently reopened to fishing exploitation whilst another area is closed. This can work seasonally or when evidence of over exploitation is presented. Often cooperation with fisherman's organisations/cooperatives is essential for the successful implementation of this management tool. For example the New Zealand rock lobster fishery uses rotational harvest as a management tool, managed between stock managers and local fisherman's organisations (Parma *et al.*, 2006). Rotational harvest has been demonstrated to be an effective tool for the management of several fisheries such as sea cucumbers (Eriksson and Byrne, 2015; O'Regan, 2015), bivalves (Hart, 2009; Kjelland *et al.*, 2015) and multi-species tropical fisheries (Cohen and Alexander, 2013).

OWF developments are often located in areas that support commercial fisheries. During their construction phase, fishing exploitation is often excluded from the site for safety reasons (Courtney French pers. comm. Senior Environment & Consents Specialist, Ørsted). Additionally, the presence of the turbines can deter certain fishing types such as mobile gear due to the practicalities of fishing in the wind farm. There are standard safety zones of 50m around each turbine and substation located in each wind farm depending on the wind farm operator (Hooper *et al.*, 2015). This can act as a protected area during the construction phase and also discourage certain fishing practices during the operation phase. Bergman *et al.* (2014) observed an increase in biodiversity and biomass of certain bivalve species within an OWF due to the absence of mobile fishing effort. There is potential for co-location of fisheries and OWFs, however these are predominantly static gear fisheries such as pot/static nets (Christie *et al.*, 2014; Hooper and Austen, 2014b; Stelzenmüller *et al.*, 2016).

4.2.1 Aims, hypotheses and objectives

The aim of Chapter 4 is to understand the effects of exclusion of fishing effort from an OWF site during the construction period and the effects of subsequent re-opening of the site and assess if an OWF site can act as a protected area.

The hypotheses to be tested in Chapter 4 were:

4.1. The size structure and catch rates of lobsters and associated commercial bycatch observed in 2015 will be positively affected by the closure of the Westermost Rough offshore wind farm development to fishing exploitation during the construction phase when compared to the control site and subsequent post-construction period.

4.2. The community assemblages derived from pot bycatch will be positively affected by the closure of the Westermost Rough offshore wind farm development to fishing exploitation during the construction phase when compared to the control site and subsequent post-construction period.

The objectives for Chapter 4 were:

- The 2015 at sea sampling data for the Westermost Rough offshore wind farm survey will be separated to reflect the period the wind farm was closed to commercial fishing exploitation (closed period) and the period during the survey once the site had been reopened to commercial fishing exploitation (open period). The control site data will be separated in the same way to assess temporal tends between the wind farm and control.
- Comparisons of size structure and catch rates of lobsters and associated commercial bycatch will be made between the wind farm and the control site during the closed period and also during the open period. Interactions between sites and the open and closed period will also be investigated.

- Comparisons of community assemblages derived from non-commercial bycatch caught in pots will be made between the wind farm and the control site during the closed period and also during the open period. Interactions between sites and the open and closed period will also be investigated.

4.3 Methods

This chapter uses the data gathered during the 2015 sampling period and used the at sea sampling protocols described in Section 2.3.2. Chapter 3 analysed the 2015 data aggregated for the whole sampling period. This chapter focuses on the 2015 data that has been separated to account for the OWF closure due to the construction phase.

Commercial fishing exploitation had been excluded from the OWF site for a total of 20 months during the construction phase (Jan 2014 – Aug 2015). The survey was permitted to sample at the pre-designated site (Figure 4.1) during the closure from 3rd July 2015. The control site was never closed to fishing exploitation.

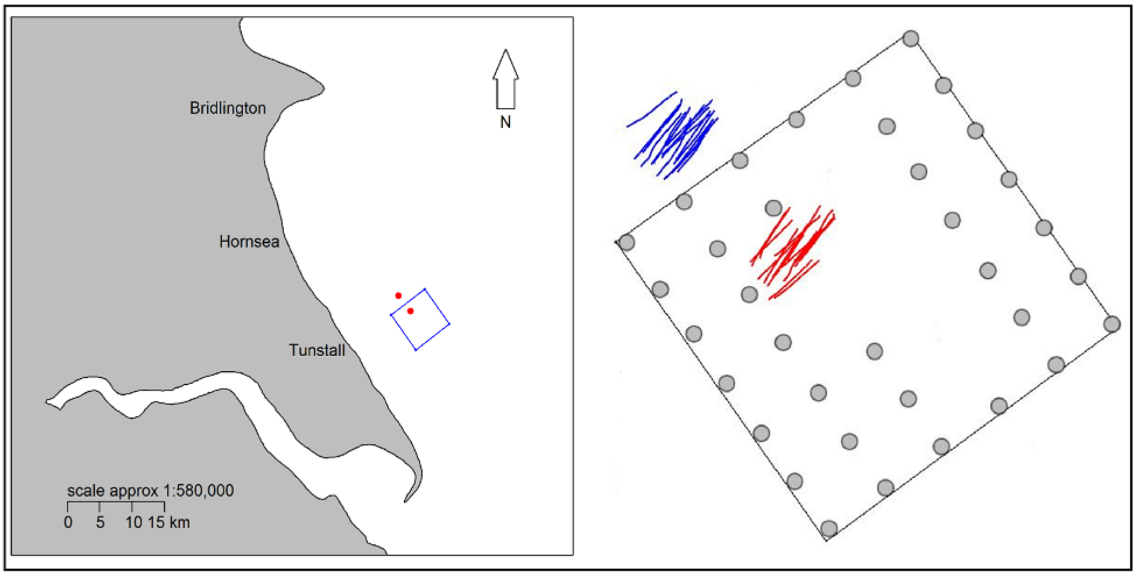


Figure 4.1: Location of the strings of pots deployed for the 2015 survey at the Westermost rough OWF.

Data was gathered from the Westermost Rough OWF site between 3rd July 2015 –and 15th August 2015 whilst the OWF site was closed to fishing exploitation (subsequently referred to as "closed"), and between 15th August 2015 and 27th September 2015, when the OWF site was reopened to fishing exploitation.

Comparisons of size structure and catch rates of lobsters and commercial by-catch were made between the OWF and control during the period the OWF was closed and also during the period once the OWF had been reopened (Figure 4.2). Testing the null hypotheses: '*Size structure and catch rates of lobsters and associated commercial by-*

catch did not differ significantly between the OWF and control site during the period when the OWF was closed to fishing exploitation and also between the OWF and control once the OWF had been reopened to fishing exploitation'.

Comparisons of community assemblages derived from commercial and non-commercial bycatch were made between the OWF and control during the period the OWF is closed to fishing exploitation and also during the period once the OWF has been reopened to fishing exploitation (Figure 4.2). Testing the null hypotheses '*Species richness, abundance, diversity and community assemblage did not differ significantly between the OWF and control site during the period when the OWF was closed to fishing exploitation and also between the OWF and control once the OWF had been reopened to fishing exploitation'.*

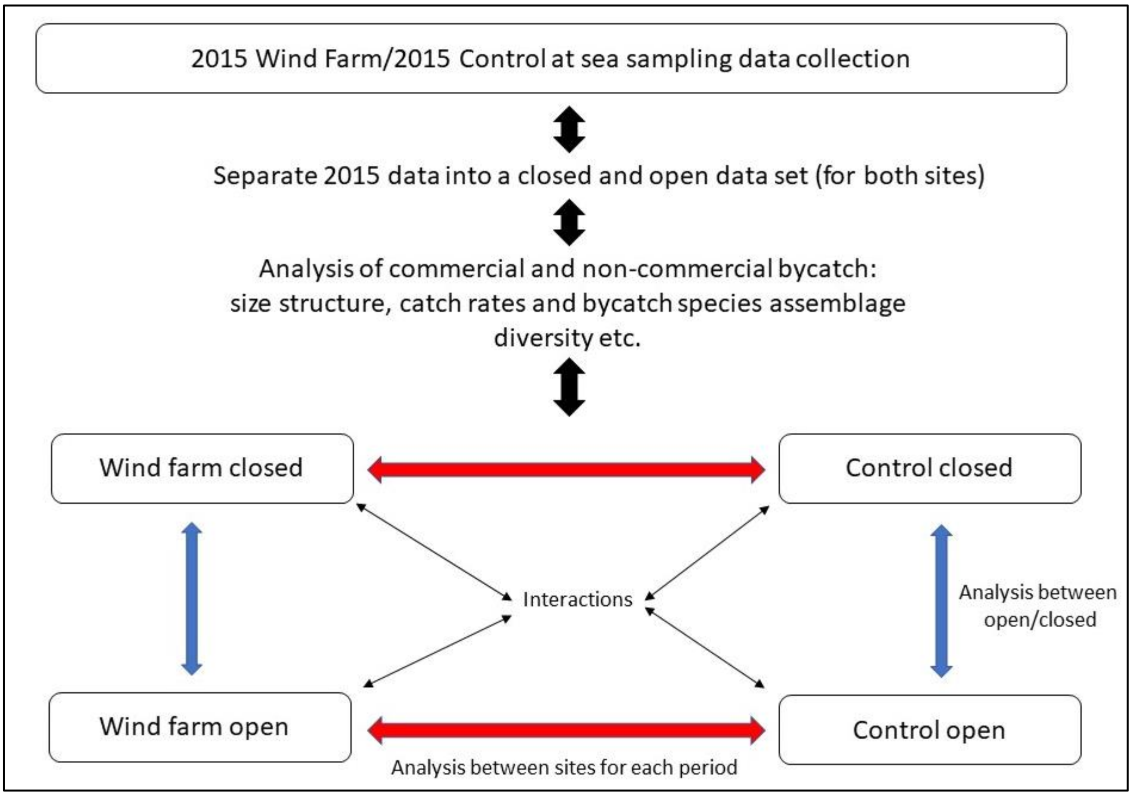


Figure 4.2: Schematic of experimental design for Chapter 4. Blue arrows denote comparison between open and closed for each site. Red arrows represent comparison between the wind farm and control sites during each period.

4.3.1 Data analysis

All analyses were for Chapter 4 were conducted using R statistical software (R Core Team, 2017) except for MDS and ANOSIM analysis conducted using PRIMER v6 (Clarke & Gorley 2006). GLMM was applied using the lme4 package in R statistical software (Bates *et al.*, 2015) and packages ggplot2 (Wickham, 2009), gridExtra (Auguie, 2017), dplyr (Wickham *et al.*, 2017), car (Fox and Weisberg, 2011), Matching (Sekhon, 2011) were used for data manipulation, analysis and graphical outputs.

4.3.1.1 Size distribution

Differences in size frequency for the lobsters and associated commercial bycatch sampled in the wind farm were compared to the control site and analysed using a two-sample K-S test. This was conducted for both the open and closed period. ECDF plots were generated to demonstrate the proportion of lobsters and commercial bycatch between each site/closure period that were less than the observed length (Ogle, 2016). A GLMM was applied in which the relative catch probability of commercial catch within each closure was the response variable, carapace length/width was the fixed effect and haul (survey day) was the crossed random intercept (Equation 1). A binomial error was applied due to the response variable being the relative catch probability of commercial catch entering pots within each year.

$$\Pr\{\frac{Test}{Test + Control}\} = 1/(1 + e^{-(\text{haul} + \beta_1 \times \text{length} + \beta_2 \times \text{length}^2)}) \qquad (1)$$

Justification and validation of the GLMM was in accordance with the protocols described in the data analysis section of Chapter 3 (3.3.1.1).

4.3.1.2 Catch comparison

Catch per unit of effort (CPUE) was determined as the total number of lobsters and commercial bycatch caught in a string (Davies *et al.*, 2015). The total number of lobsters and commercial bycatch in each string that were above their respective MLS and of good quality was determined as the landings per unit of effort (LPUE) i.e. the number of individuals that a fisher would land to market. The CPUE and LPUE data conformed to a normal distribution (S-W, $p > 0.05$). Variances could not be considered equal for lobsters (F-test, $p < 0.05$) but were considered equal for edible and velvet swimming crabs (F-test, $p > 0.05$). A Welch's *t*-test assuming unequal variances was applied to CPUE/LPUE of lobsters and a Welch's *t*-test assuming equal variances was applied to CPUE/LPUE of edible and velvet swimming crabs to analyse the difference between the wind farm and control site and also between the open and closed periods.

4.3.1.3 Non-commercial bycatch diversity

Non-commercial bycatch was determined to be an individual of any species that was not targeted intentionally and holding no market value to the fishery. All invertebrate and fish catch was classed as bycatch. The bycatch also included all commercial species such as lobster, edible and velvet swimming crabs that were below their respective MLS or above MLS but of poor quality. *Gadus morhua* (Atlantic cod) were also deemed as bycatch (even above MLS) as most of the fishery do not hold quota for whitefish.

Species richness (S), total individual abundance (n) and Shannon Weiner diversity indices (H') (LOGe) was calculated using PRIMER v6 (Clarke & Gorley 2006). and subsequent analysis (S, n & H' between sits/closure period) was conducted using R statistical software (R Core Team, 2017). The survey day/haul was used as replicates for non-commercial bycatch analysis. S did not conform to a normal distribution (S-W, $p < 0.05$), therefore a Wilcoxon rank sum test was applied to analyse if S differed significantly between sites and the open and closed period. Total number of individuals (LOGn) and H' conformed to a normal distribution (S-W, $p > 0.05$) and could be considered equal (F-test, $p > 0.05$), therefore, a Welch's two sample *t*-test was applied to analyse if LOGn and H' differed significantly between sites within each closure period and also between closure periods for each site.

MDS was applied to a Bray-Curtis similarity matrix of each of the test variables to present graphically, similarity clustering of the data. Each MDS plot was generated 999 times and the best representation presented. Stress levels below 0.1 were deemed as excellent representations and stress levels greater than 0.20 were deemed as unsuitable representations (Clarke & Gorley 2006). Analysis of similarity (ANOSIM) was conducted on all test variables (999 permutations) to statistically analyse the difference in similarity of replicates within each site and closure period.

4.4 Results

Over the 2015 survey year a total of 10182 commercial shellfish were recorded, 4619 lobsters, 3744 edible crabs and 1819 velvet swimming crabs. There were 23 survey days conducted between July and September 2015, survey days were separated into two periods; closed ($n = 12$) and open ($n = 11$). The control site which was open for the entire period was also separated into the closed and open period for comparison to account for possible seasonal fluctuations. Mean soak time for the 2015 survey was 3.9 days (s.d. +/- 2.1) with zero days omitted due to weather conditions.

4.4.1 Size distribution

During the closure period, there was a significant difference is size distribution of lobsters between the wind farm and control sites (K-S, < 0.001 Table 4.1). There was a greater proportion of smaller lobsters (< MLS) in the control site and a greater proportion of larger lobsters in the wind farm (Figure 4.3a).

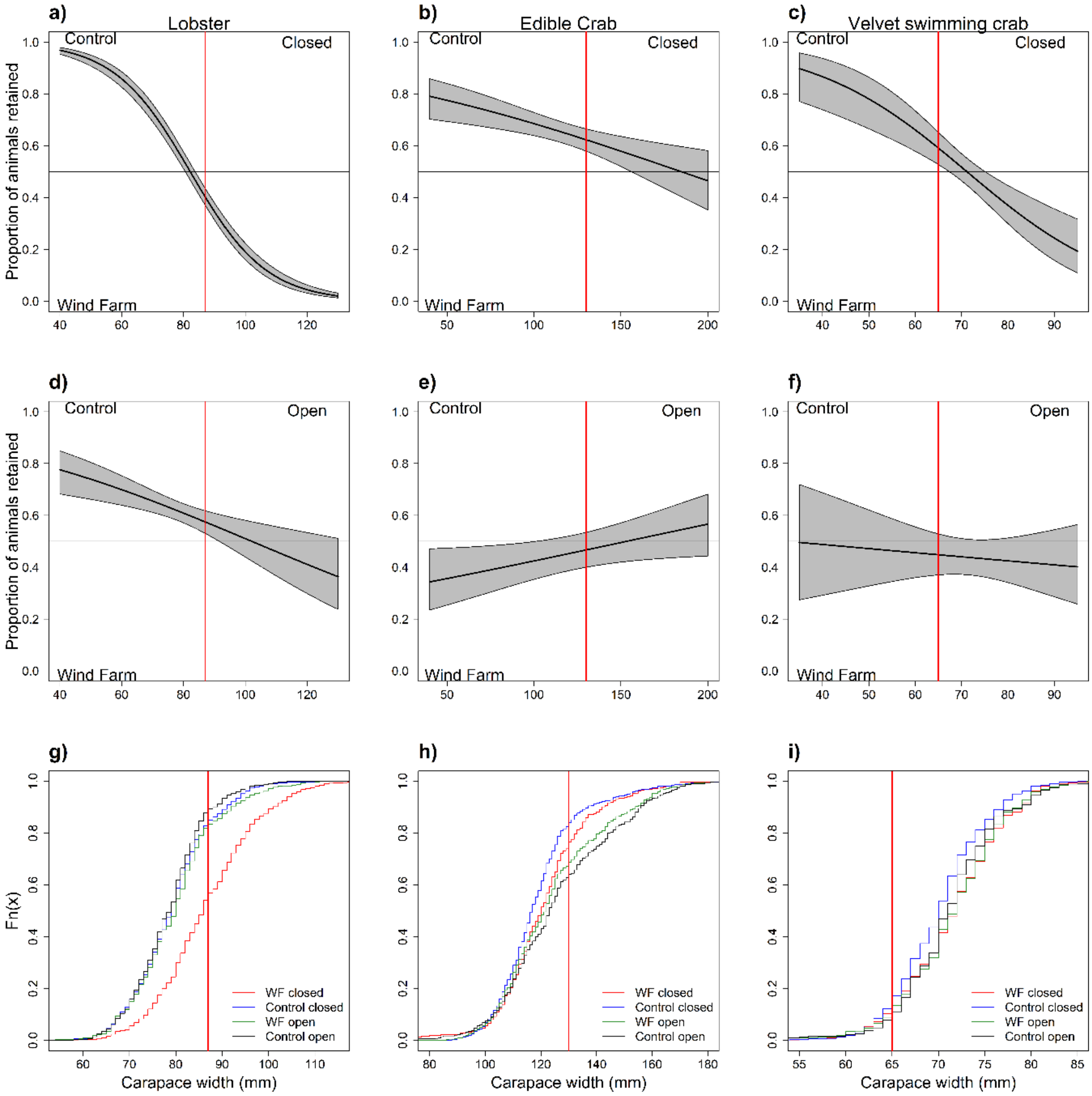


Figure 4.3: Plots derived from GLMM modelling of commercial species sampled at the WMR OWF site. a) Proportion of lobster, b) edible crab and c) velvet swimming crab in both sites during the closure period. d) Proportion of lobster, d) edible crab and e) velvet swimming crab in both sites during the open period. *The top box represents the* ***control*** *site and the bottom box represents the* ***wind farm*** *site. The grey shaded area represents the 95% confidence intervals and the bold black line the mean value. The central horizontal line represents the 0.5 (50%) values, points overlapping this line indicate that there was no significant difference in the proportion of an animal of that size between the two years. A value of 0.75 indicates that 75% of that specific species sampled at that size were sampled in 2013 and the other 25% sampled in 2015. Results of the GLMM are presented in Appendix 1, Table 4.1.*

g) Empirical cumulative distribution function (ECDF) plot of the lobster, h) edible crab and i) velvet swimming crab distribution for the **wind farm** and **control** site during the closure period (red and blue) and during the open period (grey and black). The vertical red line on all plots represents the minimum landing size of the species within the fishery. Vertical red line on all plots represents the minimum landing size of the species.

During the opening period there was a significant difference in size distribution of lobsters (K-S, p< 0.01, Table 4.1) however the proportion of lobsters within each site shifted from the distribution observed during the closure. There was still a greater proportion of lobsters < MLS in the control site during the open period, this was not as pronounced as during the closure (~25% reduction, Figure 4.3a & d). Lobsters that were greater than the MLS showed no difference in their proportion observed in each site during the open period. Lobster size distribution (Figure 4.3g) demonstrated no observable difference in the control site between closure periods and in the wind farm during the open period. During the closure period, the size distribution of lobsters in the wind farm was greater than all other cases, demonstrating a difference of up to 15 mm CL.

Edible crab demonstrated a significant difference in size distribution between the wind farm and control sites during the closed period (K-S, $p < 0.001$, Table 4.1) but was not significantly different during the open period (K-S, $p > 0.05$, Table 4.1). During the closure there was a greater proportion of edible crabs observed in the control site up to a CW of 155 mm, larger edible crabs greater than this size demonstrated no difference in proportion between the two sites (Figure 4.3b). During the open period there was no significant difference in size distribution of edible crabs between the wind farm and control (K-S, $p > 0.05$). The GLMM plot (Figure 4.3e) supports this, demonstrating only a slight difference in proportion of edible crabs, ~5% greater proportion of edible crabs up to 70 mm CW in the wind farm site. The ECDF plot (Figure 4.3h) shows that during the closure (red and blue) the size distribution of edible crabs in both sites was less than that of both sites during the open period (green and black). Edible crab size distribution was greatest in the control site during the open period (black). The distribution was approximately 20 mm CW difference between the two periods within the control site. Indicating that during the open period, across both sites there were larger edible crabs observed than during the closure period.

There was a significant difference in size distribution of velvet swimming crabs observed during the closure period between the wind farm and control (K-S, $p < 0.001$, Table 4.1). Velvet swimming crab followed a similar trend to the lobsters during the closure period. During the closure, there was a greater proportion of larger (> MLS) velvet swimming crabs in the wind farm and the inverse observed for the smaller velvet swimming crabs (Figure 4.3c). There was no significant difference in velvet swimming crab size distribution between the wind farm and control during the open period (K-S, $p > 0.05$, Table 4.1). The plot derived from GLMM (Figure 4.3f) supports this, demonstrating no observable difference between the two sites. Velvet swimming crab size distribution (Figure 4.3i) demonstrated no observable difference between closure periods in the wind farm (red and green. Size distribution of velvet swimming crabs in the control site during the closure period (blue) was less than during the open period.

Table 4.1: Results of two-sample Kolmogorov Smirnov tests, analysing the size distribution of commercial catch sampled between the wind farm and control sites at the WMR OWF between closure periods. Significant tests are in **bold**.

Variables	Status	Species	Test Statistic	p
WF > Control	**Closed**	**Lobster**	**0.33**	**< 0.001**
WF <Control	**Open**	**Lobster**	**0.08**	**< 0.01**
WF < Control	**Closed**	**Edible Crab**	**0.12**	**< 0.001**
WF = Control	Open	Edible Crab	0.07	n.s.
WF > Control	**Closed**	**Velvet Swimming Crab**	**0.16**	**< 0.001**
WF = Control	Open	Velvet Swimming Crab	0.07	n.s.

Therefore the null hypothesis *'Size structure of did not differ significantly between the OWF and control site during the period when the OWF was closed to fishing exploitation and also once between the OWF and control once the OWF had been reopened to fishing exploitation'* was rejected. The null hypothesis was accepted for commercial bycatch between the wind farm and control sites during the closed period only.

4.4.2 Catch and landings per unit of effort

Mean lobster catch per unit of effort (CPUE) did not differ significantly between the wind farm and control during the closure period (Figure 4.4a, Table 4.2). Mean CPUE reduced significantly in the wind farm site during the open period in comparison to the closure period. CPUE in the wind farm during the open period was also significantly less than in the control site during the closure period. The mean number of landable individuals per unit of effort (LPUE) was significantly greater in both the wind farm and control during the closure period in comparison to the open period (Figure 4.4b, Table 4.2). Mean LPUE was also significantly greater in the wind farm than the control site during the closure period, indicating a greater abundance of good quality, larger lobsters in the wind farm site during the closure. The greatest ratio (economic return of landings in comparison to catch) between LPUE and CPUE was greatest in the wind farm during the closure, at least 0.20 greater than other cases.

Mean Edible crab CPUE was significantly greater in the control site during the closed period than in the wind farm and the control site during the open period. Mean CPUE of edible crab did not differ significantly in the wind farm site between the open and closed periods (Figure 4.4c, Table 4.2). There was no significant difference in mean LPUE of edible crabs between either the wind farm or control and between closure periods (Figure 4.4d, Table 4.2). There was slight variation in the ratio between LPUE & CPUE (0.07 at its greatest), the greatest ratio was observed in the control site during the open period and the lowest in the control site during the closure period.

Table 4.2: Results from Welch's two tailed t-test for the mean CPUE/LPUE data analysed between the wind farm and control sites of the WMR OWF between closure periods. Significant tests are in **bold**.

Group	Species	Effort	t-value	df	p
WF = Control during closure	Lobster	CPUE	0.44	23.3	n.s.
WF > Control during closure	**Lobster**	**LPUE**	**7.09**	**15.6**	**< 0.001**
WF < Control during closure	**Edible Crab**	**CPUE**	**2.34**	**24**	**< 0.05**
WF = Control during closure	Edible Crab	LPUE	1.60	24	n.s.
WF = Control during closure	Velvet Swimming Crab	CPUE	0.23	24	n.s.
WF = Control during closure	Velvet Swimming Crab	LPUE	0.30	16.67	n.s.
WF < Control when WF open	**Lobster**	**CPUE**	**4.16**	**16.91**	**< 0.001**
WF = Control when WF open	Lobster	LPUE	0.30	17.27	n.s.
WF = Control when WF open	Edible Crab	CPUE	0.56	18	n.s.
WF = Control when WF open	Edible Crab	LPUE	0.33	18	n.s.
WF > Control when WF open	**Velvet Swimming Crab**	**CPUE**	**2.36**	**18**	**< 0.05**
WF = Control when WF open	Velvet Swimming Crab	LPUE	1.94	18	n.s.
WF between status (open<closed)	**Lobster**	**CPUE**	**4.25**	**20.42**	**< 0.001**
WF between status (open<closed)	**Lobster**	**LPUE**	**7.64**	**18.62**	**< 0.001**
WF between status (open=closed)	Edible Crab	CPUE	0.83	21	n.s.
WF between status (open=closed)	Edible Crab	LPUE	1.03	21	n.s.
WF between status (open=closed)	Velvet Swimming Crab	CPUE	2.04	21	n.s.
WF between status (open>closed)	**Velvet Swimming Crab**	**LPUE**	**-2.42**	**21**	**< 0.05**
Control between status open=closed)	Lobster	CPUE	0.25	20.84	n.s.
Control between status open=closed)	Lobster	LPUE	1.63	18.91	n.s.
Control between status open<closed)	**Edible Crab**	**CPUE**	**2.20**	**21**	**< 0.05**
Control between status open=closed)	Edible Crab	LPUE	0.01	21	n.s.
Control between status open/closed)	Velvet Swimming Crab	CPUE	0.27	21	n.s.
Control between status open/closed)	Velvet Swimming Crab	LPUE	-0.04	21	n.s.

Mean CPUE of velvet swimming crabs during the open period was significantly greater in the wind farm than the control and was also significantly greater than the wind farm during the closed period (Figure 4.4e, Table 4.2). There was no significant difference in mean LPUE of velvet swimming crabs between the wind farm and control and between the open and closed periods (Figure 4.4f, Table 4.2). The greatest ratio of LPUE to CPUE was observed in the wind farm for both closure periods. Although LPUE was less in the wind farm during the closure, it had the same ratio as during the opening, indicating a higher economic per unit of effort return during the closure period.

Therefore the null hypotheses 'C*atch rates of lobsters and associated commercial by-catch did not differ significantly between the OWF and control site during the period when the OWF was closed to fishing exploitation and also once between the OWF and control once the OWF had been reopened to fishing exploitation'* was rejected.

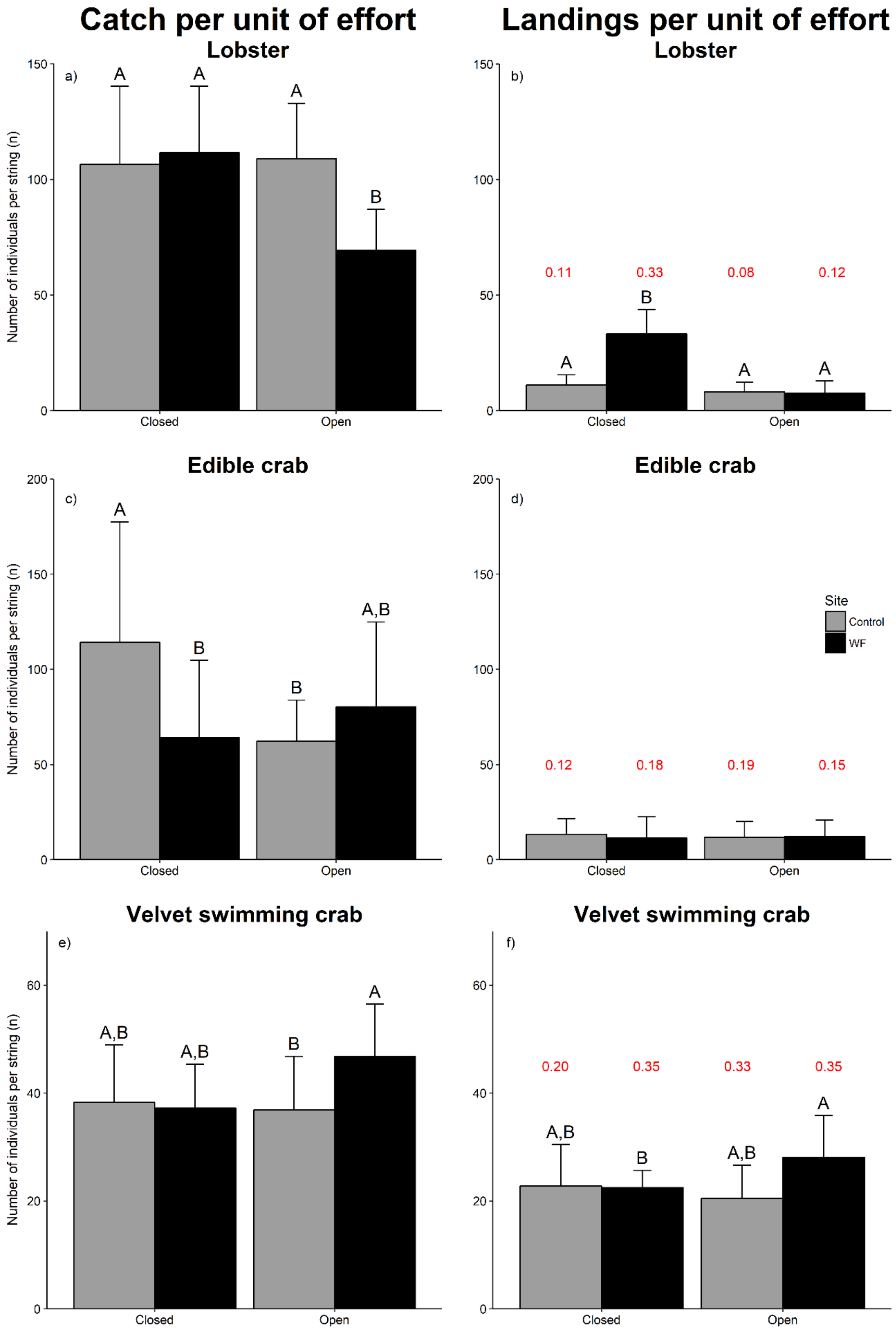


Figure 4.4: Bar plots of catch per unit of effort (CPUE) for lobster and the number of landable individual per unit of effort for lobster (LPUE) a) & b), edible crab c) & d) and velvet swimming crab e) & f), sampled between sites and statuses of the WMR OWF. The top of the bars represents the mean value of y and the top of the error bars represent the standard deviation of y; this applies to all subsequent bar plots reported.

4.4.3 Diversity

There was no significant difference in species richness (S) between the wind farm and control sites or between the open and closed period (Figure 4.5a & c, Table 4.3). Mean species abundance (logN) was significantly greater in the control site during the closure period than the wind farm site during both the open and closed period (Figure 4.5, Table 4.4). There was no significant difference in mean species abundance (logN) in the wind farm or control site during both the open and closed period and no significant interaction between site and closure period (Tukey, $p > 0.05$, Table 4.4).

Table 4.3: Results of Wilcoxon rank sum analysis of species richness (S) between the wind farm and control sites and closure periods.

Site	Variable	Test	Test statistic	df	p
Control between closures	S	Wilcoxon rank sum	80.0	n/a	n.s.
WF between closures	S	Wilcoxon rank sum	87.5	n/a	n.s.
WF and Control during closed period	S	Wilcoxon rank sum	57	n/a	n.s.
WF and Control during open period	S	Wilcoxon rank sum	56	n/a	n.s.
WF Open = Control Closed	S	Wilcoxon rank sum	74	n/a	n.s.
WF Closed = Control Open	S	Wilcoxon rank sum	93	n/a	n.s.

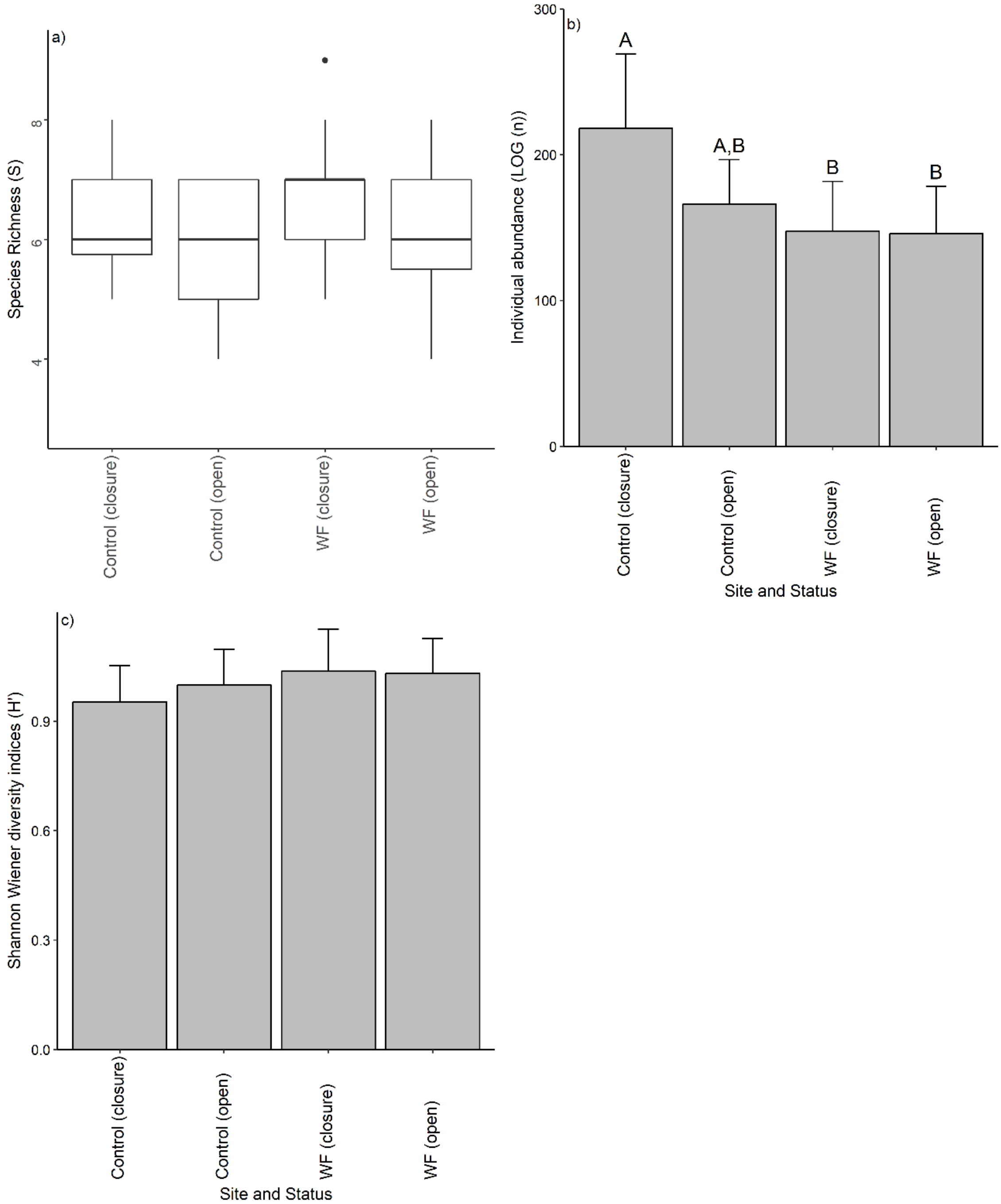


Figure 4.5: a) Boxplot of species richness for both the control and wind farm sites for each period of closure. *The median of the value of y is represented by the bold horizontal line; the top of the box above the median represents the 75th percentile with the bottom of the box representing the 25th percentile. The whole box represents the interquartile range (25-75), the whiskers represent the maximum and minimum values of y. The outliers represent data points that were greater than 1.5 times the interquartile range above the third quartile or below the first quartile. This applies to subsequent box plots reported in this study.* b) Bar plot of the mean s number of individuals (species abundance) and c) Shannon Weiner diversity indices recorded at both the control and wind farm sites for each period of closure.

Mean Shannon Weiner diversity indices (H) did not differ significantly between the wind farm and control sites and also between the open and closed period. There was also no significant interaction between site and closure period (Figure 4.5, Table 4.4).

Table 4.4: Results of two-way ANOVA analysing species abundance (logN) and Shannon Wiener diversity indices (H) between sites and also between the open closed period. Results of Tukey test presented.

Variable	Factor	Sum Sq	F value	p	Differences (Tukey test)
logN	Site	0.8251	15.72	< 0.0001	Control closed > WF open & WF closed
	Closure	0.2115	4.03	n.s.	n.s.
	Site*Closure	0.1768	3.37	n.s.	n.s.
H	Site	0.402	3.806	n.s.	n.s.
	Closure	0.0046	0.439	n.s.	n.s.
	Site*Closure	0.0075	0.715	n.s.	n.s.

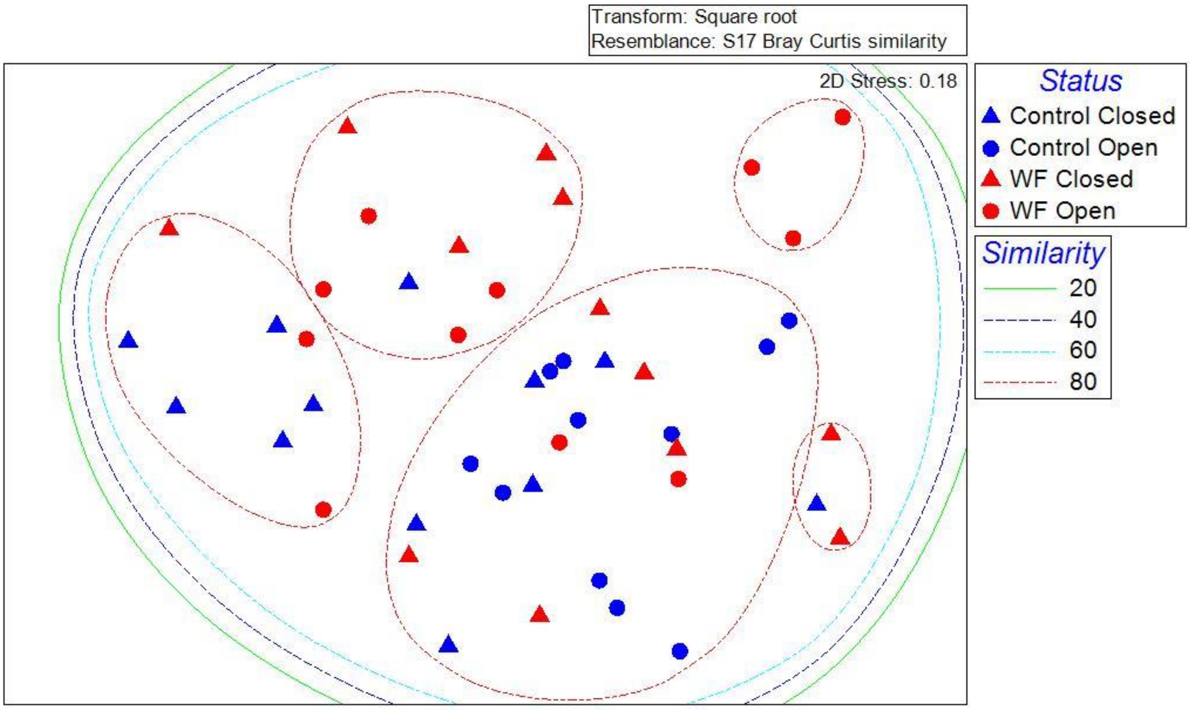


Figure 4.6: The results of MDS analysis on a Bray-Curtis similarity matrix of the bycatch communities observed in lobster pots sampled at the WMR OWF site. The status data represents both the Wind Farm (red) and the Control Sites (blue) and the status of the wind farm (open/closed). There was general overlap of points between the wind farm and control during the closed period (triangular points). The same was observed during the open period (circular points). All sites/status fall within a 60% similarity. There were 5 clusters of points representing 80% similarity. The stress value of the 2D plot is 0.18 indicating that this was a reasonable representation of the similarity between sites and status of the wind farm.

There was a significant difference in the bycatch pot communities sampled during the closure period at the WMR OWF site (ANOSIM, Global R = 0.131, p = 0.1%, Figure 4.6).

Bycatch pot communities differed significantly in all cases except for the wind farm site between closure periods (Table 4.5). Similarity clustering (Figure 4.6) highlighted 5 clusters (80% similarity), two large clusters with points from both sites and the open closed period and three separate clusters which were dominated by points representing the wind farm. However, all clusters/points had a similarity of 60%.

Table 4.5: Results of ANOSIM on Bray-Curtis similarity matrix of the communities sampled in lobster pots at the WMR OWF site between wind farm and control sites between closure periods. Significant variables are in **bold**.

Group statistics	**Global R**	***P (%)***
Control Closed / Control Open	**0.238**	**0.2**
Control Closed / WF Closed	**0.119**	**3.5**
Control Closed / WF Open	**0.156**	**2.2**
Control Open / WF Closed	0.083	7.6
Control Open / WF Open	**0.192**	**0.6**
WF Closed / WF Open	0.017	32.9

Non-commercial bycatch (below their respective MLS or > MLS but of poor quality) of lobsters, edible crab and velvet swimming crab were the dominant species accounting for similarity in community assemblages between the wind farm and control during both the closed and open period. (Table 4.6). Non-commercial edible crab bycatch contributed the greatest to the dissimilarity between the wind farm and control sites during both the open and closed period (Table 4.6).

Table 4.6:Results of SIMPER analysis, highlighting dominant species contributing to the similarity between sites and the open and closed period (90% cut off for low contributors).

	Lobster	Edible crab	Velvet swimming crab
Average abundance WF closed	73.75	54.25	14.42
Average abundance WF open	61.73	63.00	16.36
Average abundance Control closed	93.5	104.75	N/A
Average abundance Control open	97.91	48.64	N/A
% contribution to dissimilarity Control closed ~ Control open	30.70	57.44	4.97
% contribution to dissimilarity Control closed ~ WF closed	33.06	55.34	5.57
% contribution to dissimilarity Control open ~ WF open	42.22	42.64	7.53
% contribution to dissimilarity WF closed ~ WF open	33.48	49.00	8.89

4.5 Discussion

4.5.1 Size distribution

The closure of the WMR OWF during the construction phase and subsequent exclusion of fishing effort was found to influence the size distribution of lobsters and commercial bycatch in the area. This closure and subsequent exclusion of fishing effort acted as an NTZ for the closure period. This effect protected the lobsters greater than the MLS in the wind farm from exploitation. This increase in the size spectrum of lobsters can have the inverse effect on edible crabs due to inter-species competition (see Figure 2.12). This was observed during the closure period where there was a greater proportion of larger lobsters in the wind farm that was not observed for larger edible crabs. Many crustacean species are size dominant, with larger animals asserting dominance over smaller animals of the same species (Phillips, 2007).The greater presence of larger lobsters may have deterred smaller lobsters and commercial bycatch from either the site or entering the survey pots (Jury and Watson, 2013). The reduced proportion of smaller lobsters within the closed area indicate that these size classes of lobsters may have been displaced from the site. Thorbjørnsen *et al.* (2018) observed that there was overspill of larger lobsters from an MPA into surrounding fisheries areas, however this was not the case in this study. The temporal scale of the current study may not have been sufficient to observe this effect, however displacement of smaller lobsters was observed. Jury and Watson (2013) observed that crab species were quicker than *H. americanus* to encounter and occupy a pot in a mixed crustacean fishery, therefore it was expected there was a greater proportion of crabs in the wind farm during the closure period as they were not being exploited. As this was not the case in the present study, it can be assumed that the greater proportion of larger lobsters observed in the wind farm had an influence on the edible crab population. The greater abundance of larger lobsters in the site were also potentially occupying the pot before the edible crabs, deterring them from entering. Velvet swimming crabs followed the same trend as the lobsters, demonstrating a greater proportion in the wind farm during the closure period. This greater proportion was as expected due to the fishing pressure being absent during this period. The inter-species interactions between lobsters and crab species (as observed with edible crab (see Figure 2.12) may not be as detrimental to velvet swimming crabs, allowing for occupation of similar niches.

Once the wind farm had been opened to fishing exploitation, the site was exposed to high levels of fishing effort, leading to a rapid, short-term increase in landings from the site (2200 kg (day prior to opening) increased to 4490 kg landed the day immediately post opening of the site (data provided by Bridlington Shellfish and ISC Bridlington)). This intensive effort on commercial catch greater than MLS, altered the size distribution of the previously unfished population over a relatively short period. However, for lobsters this reduction in proportion at size, did not decrease below that observed in the control site. There was still a greater proportion of lobsters below MLS observed in the control site post opening. The removal of lobsters greater than MLS could reduce the displacement of smaller lobsters from the wind farm site. A reversal of this effect may not have happened during the period of this survey. The opening of the wind farm and intense exploitation of larger lobsters may have had a positive effect on the proportion of edible crabs between the wind farm and control, reducing the inter-species competition associated with pots. The shift in size distribution of edible crabs, indicated only a slight difference in the proportion of edible crabs retained at each size between the two sites. The increased proportion of smaller edible crabs and absence of larger lobsters in the wind farm post-opening, indicates that the lobsters were displacing the smaller edible crabs from the area or deterring them from entering the pots. The size distribution of velvet swimming crabs between sites, post opening, not being significant suggests that the lack of fishing exploitation was the cause of the difference when the wind farm was closed. However, these results should consider that the catch dynamics between lobsters and commercial bycatch species makes it difficult to ascertain the effects of the closure and subsequent reopening of the OWF site on the size structure of commercial bycatch species.

4.5.2 Catch and landings per unit of effort

The closure of the wind farm site influenced the CPUE of the three commercial species sampled. Although lobster CPUE was greatest in the wind farm during the closed period it was not significantly different to the control site. Hoskin *et al.* (2011) observed an increase in abundance of larger lobsters in the Lundy Island NTZ due to the absence of fishing exploitation. The increase in size of lobsters in the wind farm and the non-significant difference in CPUE between sites observed in this study, supports this theory. A general increase in larger lobsters but not a greater abundance when compared to the control site. This was also reflected in the significantly greater LPUE in the wind farm during the closure. This indicates a greater abundance of good quality lobsters above MLS during the closure. Opening of the wind farm to fishing exploitation saw a significant reduction in CPUE of lobsters between the two sites and a significant reduction of LPUE within the wind farm between the open and closed period. However, there was no significance between the wind farm and control sites during the open and closed period, indicating that the economic return for fishers in the wind farm reflected that of the control site within a relatively short period.

Edible crab CPUE was significantly greater in the control site than the wind farm during the closure, and significantly greater in the wind farm than the control site during the open period. There was no significance difference in edible crab CPUE in the wind farm between the open and closed period. Displacement of edible crabs from the wind farm site into the control area during the closure period may account for this increase. The NTZ effect of the closed area may also be producing spill over of certain species. It has been demonstrated that MPAs/NTZ can benefit surrounding areas by protecting stocks within the boundaries and generating spill over into the surrounding areas (Freeman *et al.*, 2009; Pettersen *et al.*, 2009; Moland *et al.*, 2011b; Huserbråten *et al.*, 2013; Thorbjørnsen *et al.*, 2018b). However; Turner *et al.* (2013) observes that there is potential for increased fishing effort around closed areas due to displacement of effort. With static fisheries there is potential to have increased effort in an area due to "effort squeeze", a process where displacement from one area forces more fishing gear into another. This can be due to limitations of the vessel, limited fishing ground supporting adequate yield and a fisher's fidelity to specific sites. As the control site was subjected to fishing exploitation throughout the survey, it is more likely that the seasonal trends in the edible crab fishery were responsible. Edible crabs in the North Sea undergo ecdysis in the warmer months in order to breed. Öndes *et al.* (2017) observed a greater proportion of male edible crab discards due to ecdysis during the summer months. Ovigerous crabs are rarely observed in pots (during this survey none out of 1357 female edible crabs were ovigerous) due to the fact they rarely move or feed during the brood period (Howard, 1982). The open period was after the middle of August 2015, this may have coincided with the moult period and influenced the reduction in CPUE. The closure and subsequent reopening of the wind farm had no effect on the LPUE of edible crabs, indicating that although CPUE was affected by the closure, the fisher would not benefit from any potential over spill effects with regards to edible crab.

Both CPUE and LPUE of velvet swimming crabs was greatest in the wind farm during the open period. Krone *et al.,* (2017) observed an increase in edible crabs on the base of turbines fitted with scour stone protection. It is possible, due to displacement of edible crabs by lobsters from the wind farm site, the velvet swimming crabs may have occupied this niche in the absence of edible crabs, enhancing their population. As the increased CPUE/LPUE of velvet swimming crabs was during the open period this is unlikely, it is more likely that the velvet swimming crabs took advantage of the increased exploitation of lobsters immediately post opening of the wind farm. Disturbance can benefit some marine organisms (Ramsay *et al.*, 1998), velvet swimming crabs may have benefited from the disturbance caused by the construction of the wind farm and possible displacement of other decapod species due to the construction process.

Whilst there was positive responses to the size and catch rates of lobsters in the wind farm during the closure period, this was not the same for the commercial bycatch in the site, therefore Hypothesis 4.1 was partially accepted due to the positive effects only observed in one of the study species.

4.5.3 Diversity

MPAs have been demonstrated to improve biodiversity in a variety of marine habitats (Jones *et al.*, 2007). In this study, species richness and diversity showed no significant difference either between sites or closure period. Abundance of individuals was significantly greater in the control site during the closure than the wind farm site during both the closed and open period. This could be attributed to the increased CPUE of edible crabs in the control during the closure, however this was not at a level to offset the other sites/periods. The non-commercial bycatch may account for this increased abundance of individuals; however, these were observed in low abundance across all replicates. Although there was little overall separation of points within the MDS for the control site, ANOSIM demonstrated a significant difference in bycatch communities between closure periods of the control sites. Only the wind farm site between the open and closed period and the wind farm (closed) and control (open) demonstrated no significant difference in bycatch communities. Demonstrating that the closure of the wind farm site had no observable effects on the non-commercial bycatch assemblages in the wind farm site, therefore Hypothesis 4.2 was rejected. The presence of artificial structures in the marine environment can act as a fish aggregation device (FAD) (Griffin *et al.*, 2016). The presence of the monopiles/scour stone may be acting in this capacity, thus increasing diversity. However, species associated with FAD's use them for shelter, predator avoidance and foraging, they tend to be spatially restricted to the individual structure with little transit between (Griffin *et al.*, 2016). The likelihood of encountering the survey pots was slim as there was a 50 m safety zone around each turbine so the survey strings were at least 50 m from each structure. The dominant non-commercial species were edible crab, lobster and velvet swimming crab, accounting for over 90% of the similarity between treatments. The lack of a significant difference in bycatch communities in the wind farm between closures indicates that the bycatch communities were not affected by the presence or absence of fishing exploitation. The results of the diversity analysis should be taken into context with the sampling method used. The results show changes in bycatch communities, i.e. species that are incidentally sampled in pots, not species that will not be sampled using this method. For a true reflection of the influence of closed areas due to wind farm construction on marine communities a targeted survey with correct sampling techniques would need to be conducted.

4.5.4 Wind farms as closed areas

There can be detrimental effects on crustacean fisheries to permanently closed areas. For example, increased inter and intra-species competition has been observed to lead to increased damage and spread of disease in lobsters (Wootton *et al.*, 2012; Davies *et al.*, 2015), and an increase in dominance of one species such as lobster, reducing overall diversity (Hoskin *et al.*, 2011). Thorbjørnsen, *et al.* (2018) observed that although there was an increase in emigration of lobsters from an MPA to a surrounding fished area, this was just an increase in overall size not abundance of lobsters but did not report on disease or injury. Closed areas and the concept of rotational harvest are often used as a fisheries management tool (Hart *et al.*, 2002; Cohen and Foale, 2013a; Eriksson and Byrne, 2015; Kjelland *et al.*, 2015; O'Regan, 2015). Closed areas can be most effective on species that have a sessile lifestyle or limited home range. Lobsters have been proven to have a limited home range within a season (Smith *et al.*, 1998) but do however have a seasonal migration offshore. Closure of areas that support important crustacean fisheries can protect the spawning stocks (individuals over MLS) which are fitter and generate a greater quantity of more robust larvae (Tully *et al.*, 2001; Goni *et al.*, 2003; Émond *et al.*, 2010; Moland *et al.*, 2010). However, it has been suggested that closed areas for the purpose of crustacean fisheries management have a finite life, suggesting closure for an optimum period to enhance ecological benefits but reopening prior to potential detrimental effects being observed (Davies *et al.*, 2015; Roach *et al.*, 2018). Closed areas should only be implemented for the right reasons (Kaiser, 2005; Caveen *et al.*, 2014) and it has been argued that MPAs do not offer sufficient protection for conservation species (Costello and Ballantine, 2015). The development of OWFs, via their exclusion of fishing exploitation during their construction phase can act as quasi-NTZ/MPA. Permanent closure of OWF sites may be detrimental to local fisheries. Stelzenmüller *et al.* (2016) predicted that there would be a 50% loss to a static netting fishery in the German Economic Zone, however this was not the case for a static potting fishery in the same region.

There is potential for OWFs, with their easily defined boundaries, to be used as a crustacean stock management tool. Combined with other suitable sites, rotational closures could protect spawning stocks whilst subsequent reopening could offset economic loss to the fishers.

4.6 Conclusions

This study has demonstrated that exclusion of fishing in a wind farm site has had a positive effect on size, CPUE and LPUE of the lobster stock in comparison to the control site which was exposed to fishing exploitation throughout. However, this closure whilst benefiting the lobster population, saw a reduction in the catch rate of edible crabs. Subsequent reopening of the site saw a shift towards a smaller size of all three commercial species reflecting that of the control site. Exploitation levels immediately following the opening of the wind farm were high, however the LPUE quickly reflected the control site, indicating a similar LPUE for a fishers' effort in a short period.

Seasonal or rotational closures, accounting for temporal and ecological factors can protect spawning stocks at optimum periods. Subsequent reopening of the site can offset potential economic loss to the fishers and may reduce the displaced fishing effort due to the closure. However, care needs to be taken to avoid the "race to fish", where the fishers that are most capable of taking advantage of a previously closed area, may not be those that were impacted most by the closure.

Chapter 5: Assessment of methods for determining egg number of the European Lobster (*Homarus gammarus* (L)).

5.1 Abstract

Fecundity estimates (reproductive potential) are essential to fisheries managers for estimating yield per recruit and the reproductive potential of a stock. However, there is no standardised methodology for determining egg number in *Homarus gammarus* stocks. A methodology was needed in to estimate egg yield from a population. Regression analysis was used to assess the accuracy of three common methods (wet/dry weight and a visual method) for estimating egg number relative to actual counts determined by an automatic egg counter. Morphological characteristics (carapace length and width and width of abdominal segments) and a proxy for egg number (P2) were investigated as suitable predictors for egg number. Carapace length, the industry standard, was found to be a poor predictor of egg number across all methods trialled. Stepwise regression was used to determine the best morphological variable or combination of such to predict egg number. There was no significant difference in the regression models generated. Based on acceptable error tolerances, the dry weight method was the most suitable method to predict egg number when combined with the width of the fifth abdominal segment or the dry weight of the egg proxy (P2). This chapter has compared the common methods for determining egg number in *H. gammarus*, and generated models for best accuracy. It has also discounted estimates by eye as a valid method for determining fecundity in *H. gammarus.*

5.2 Introduction

Fecundity estimates (i.e. the number of eggs a female produces) are used in management of marine resources such as commercial fisheries for fish and shellfish. Eggs are extruded from the female to attach to ovigerous setae on the pleopods on the ventral side of the abdomen, not all eggs extruded are fertilised, this can be due to the female lobster having not mated, is sexually immature or has mated with a sterile male (Talbot *et al.*, 1984a; Watson *et al.*, 2017) Fecundity estimates can be used to determine reproductive outputs from a stock (Laurans *et al.*, 2009) or used to generate yield per recruit models (Smith & Addison, 2003). Fecundity has been shown to be directly linked to maternal size in decapods such as lobster (Agnalt, 2008; Ellis *et al.*, 2015; Linnane, Penny, & Ward, 2008), crabs (Tallack, 2007a; Tallack, 2007b; Ungfors, 2007) and shrimps (Bilgin & Samsun, 2006).

Fecundity estimates in marine invertebrates such as *H. gammarus* are difficult to ascertain due to the large numbers of eggs involved. A variety of methods have been applied in assessing fecundity in lobster species (

Table 5.5). This lack of standardisation makes it difficult to compare fecundity estimates from different sources and geographical regions (Lizárraga-Cubedo *et al.*, 2003; Ellis *et al.*, 2015). The most common methods involve stripping and weighing the egg mass (either wet or dry) and then counting and weighing a subsample of eggs to determine overall egg number. The two most common methods used to determine egg number in *H. gammarus* are wet weight (as described by Agnalt (2008)) and dry weight (as described by Lizárraga-Cubedo *et al.*, (2003))

Non-invasive techniques have been used such, as estimating egg mass volume whilst still attached (Currie *et al.*, 2010, Coleman *et al.*, 2019). This requires removal of a small proportion of eggs, calculating the volume of a single egg and estimating the number of eggs in a brood depending on the volume of the egg mass. Another potential method involves the use of images of the ventral abdominal surface of ovigerous lobsters where the egg number had previously been determined, via invasive techniques, as a guide to estimate fecundity (Talbot *et al.*, 1984). However, these methods are initially invasive and involve egg removal to generate the models for a non-invasive technique. Initial validation of the methodology would be needed for fecundity studies for specific species/populations. Non-invasive techniques allow large numbers of lobsters to be sampled without removing the eggs from the animal, causing little stress, and allowing the eggs to remain. These methods, whilst allowing large numbers of replicates has the potential for introducing inaccuracy in the estimates, therefore the non-invasive techniques are not currently used as widely as invasive techniques. However, recent non-invasive techniques reported <1.5% difference in fecundity estimates derived from estimating egg number based on brood volume when compared to the traditional dry weight method (Coleman *et al.*, 2019). The use of non-invasive techniques may be encouraged by managers, with the introduction of the ban on landing ovigerous lobsters in 2017 preventing ovigerous lobsters being landed (DEFRA, 2017).

The accuracy required for egg number estimates, the resources available and the time taken to gather the estimates can influence the method used.

5.2.1 Aims, hypotheses and objectives

The aim of Chapter 5 is to generate a model for predicting egg number, based on the most suitable morphological characteristics of the female lobster (carapace and abdomen dimensions) and using the most suitable methodology for estimating egg number (wet weight, dry weight and visual estimations).

The hypotheses to be tested in Chapter 5 were:

5.1. Comparison of different methods for determining egg number (wet, dry, and visual) in lobsters will highlight the most suitable method to use when compared to an actual count.

5.2. Carapace length of lobsters will be a suitable predictor for determining egg number in lobsters irrespective of the method used to estimate egg number.

5.3. Combining morphological characteristics will produce a model with greater accuracy than carapace length alone when predicting egg number in lobsters.

The objectives of Chapter 5 were:

- Estimates of egg number from a range of lobster sizes will be made using three different methods: wet, dry and a visual method. These will be compared to an accurate estimate generated by an automatic egg counter.
- Carapace length will be used to predict egg number based on the three different methods (wet, dry, and visual) and assessed for accuracy.
- Models will be generated to assess the suitability of using different morphological characteristics, including a proxy for egg number, in estimating egg number in lobsters.
- Models will be assessed for accuracy in relation to an acceptable error tolerance (what level of error is acceptable in the egg number estimate).

5.3 Methodology

5.3.1 Sampling Regime

There is potential for egg loss seasonally and during handling and transportation of berried lobsters (Lizárraga-Cubedo *et al.*, 2003), this is often the case when sampling at the quayside. To avoid this, all lobsters were sampled directly from the lobster pots during routine sampling on board the *R.V. Huntress* in July and August 2014. On hauling, ovigerous lobsters were separated from the main catch and held individually ready for egg removal. To investigate whether a proxy could be determined, eggs attached to the second pleopod, were removed and preserved separately to the rest of the egg mass (subsequently referred to as P2) (Figure 5.1). All eggs were removed by hand and immediately put into a 4% formol saline solution for preservation (Tully *et al.,* 2001), whilst preservation can affect the weight of the eggs, it should affect the weight of the eggs equally, therefore not influence egg number estimations. Any eggs that were not removed were counted (including P2) and added to the total egg number.

Figure 5.1: Ventral view of an ovigerous *H. gammarus*. Highlighted is the first pair of fully formed pleopods (P2), these eggs were separated to investigate a potential proxy for determining fecundity.

5.3.2 Methods for determining egg number

Three different proxies for egg number were used to compare with a precise count taken using an automatic egg counter (Figure 5.3). These were wet weight, visual estimation, and dry weight. Subsequently referred to as wet, dry and visual. Egg samples were removed from the lobster and a visual estimate of egg number recorded, the sample was wet weighed and egg number estimated by wet weight. The individual eggs within the egg mass were separated and counted using an automatic egg counter to give an actual count. The sample was dried, and an egg number estimate calculated using dry weight (Figure 5.2).

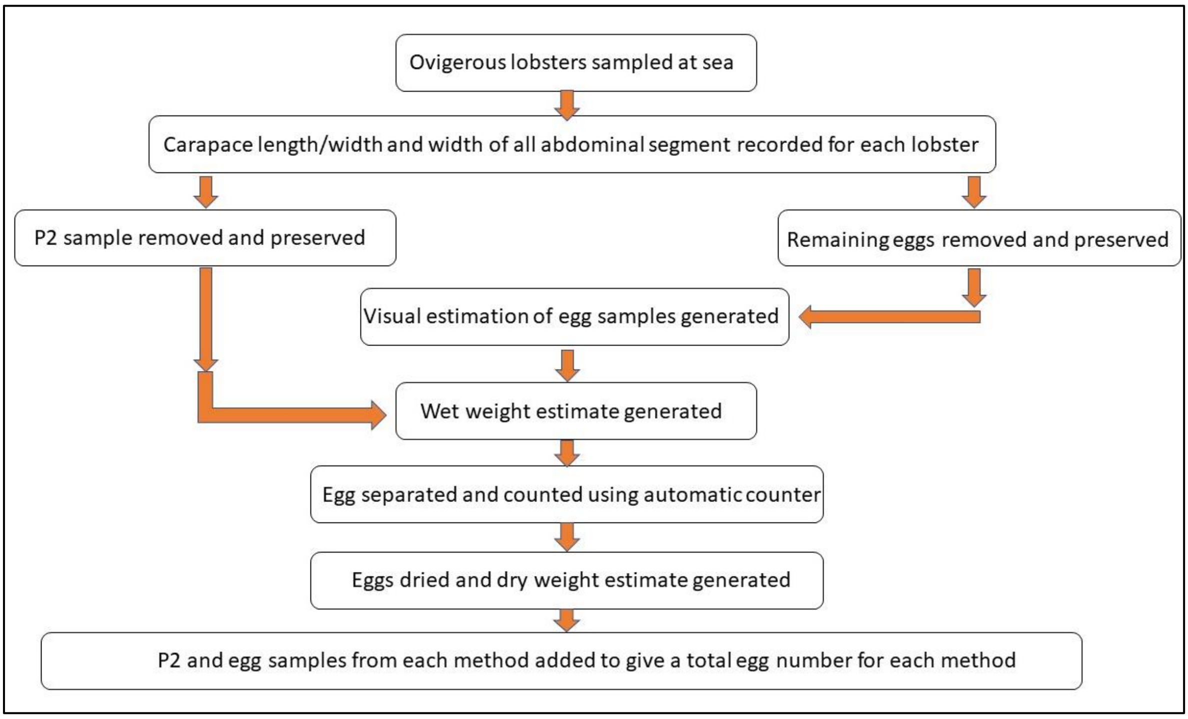


Figure 5.2: Schematic of experimental design for Chapter 5. The process for gathering and processing egg samples. Each egg sample from a lobster followed the same process.

5.3.2.1 Visual estimation of egg number

It is possible for an experienced observer to estimate egg number based on experience (Holden & Ellner, 2016). However, the level of experience of observers can differ greatly. Using a visual guide can greatly increase the estimate of observers. In the field the use of images of the ventral side of berried lobsters with a known number of eggs can be used as a visual guide for observers. Talbot *et al.*, (1984) used photographs in the field of the ventral side of ovigerous lobsters with various known quantities of eggs to act as a standard for estimating egg number. To replicate the potential for in-field visual estimations, eggs were placed in 60 ml sample pots and the egg samples were compared to known numbers of eggs in the same size pots. Whilst this does not truly reflect estimating egg number in the field it acted as a proxy for a visual estimation. The visual guide pots held a range of eggs of known quantities and different pots held eggs of different development stages. The visual guide pots contained eggs numbering 1000, 2500, 3000, 5000 and 8000. The visual guides and the samples were given to observers with different levels of experience, these included the author, academic staff and post-graduates practised in ecological estimations and undergraduates with little to no experience. Observers worked independently and were permitted to move back and forth between samples to try to improve their estimates and there was no time limit set. The estimations were made to the nearest 500 eggs in most cases.

5.3.2.2 Wet Weight

Eggs were removed from the preserving fluid and all excess moisture removed by placing the eggs on a paper towel for 30 minutes. Egg mass was weighed using an Adventure Pro AV114C fine balance scale to the nearest milligram. For each sample, a small proportion of eggs were removed and counted, each sample was counted by eye three times and a mean taken to account for counting errors (Agnalt, 2008). The sub sample was weighed and egg number for the entire egg mass calculated using:

$$N = tw * (sn/sw) \tag{2}$$

where N represents the number of eggs, tw represents the total wet weight of the egg mass, sn represents the number of eggs in the sub sample and sw represents the wet weight of the sub sample.

5.3.2.3 Automatic Egg Counter

To facilitate the use of the automatic egg counter, the eggs were separated by dissolving the funiculae (bonds between eggs) using a 5% sodium hypochlorite, household bleach solution in sea water (Choy, 1985). The use of sea water in the solution reduces the chance of egg rupture due to the permeability of the egg membrane to fresh water.

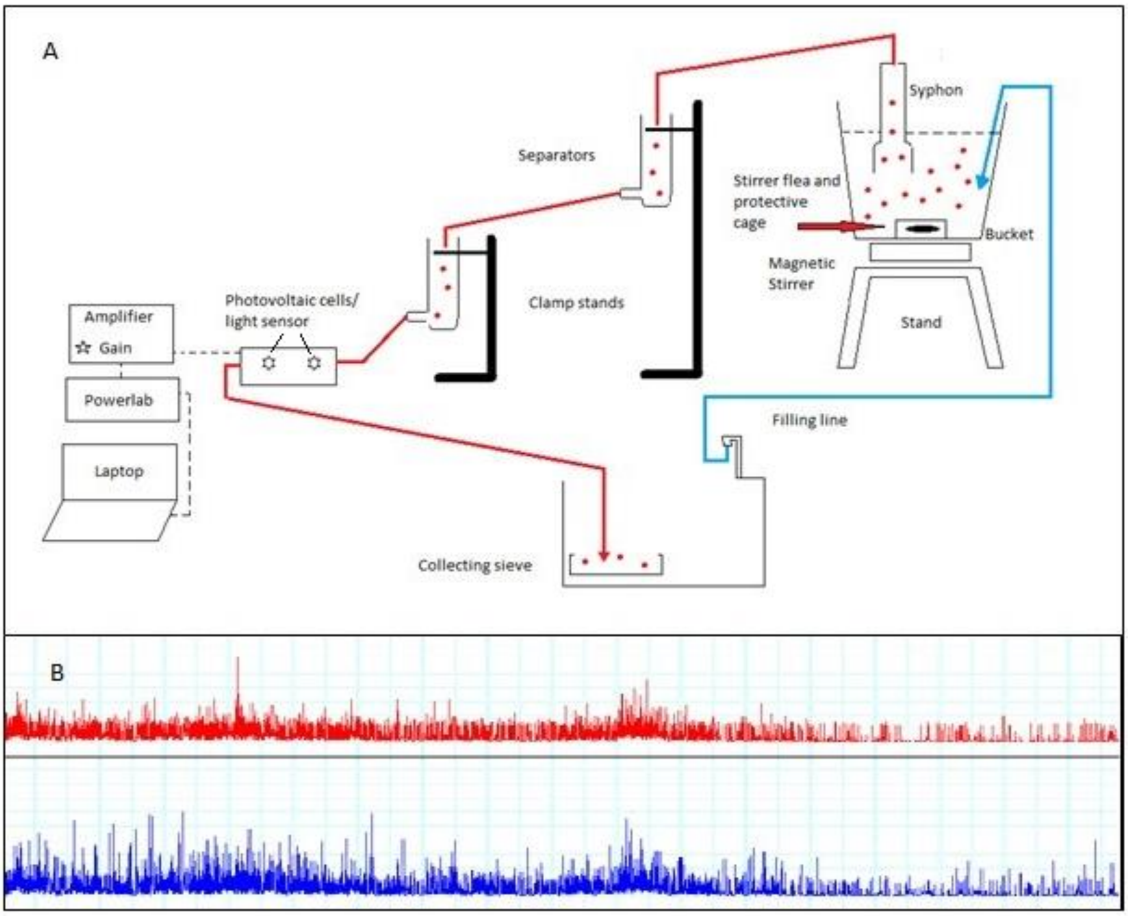


Figure 5.3: Schematic of automatic egg counter and example trace. a) The automatic egg counter used to count separated egg samples. Eggs are suspended in the bucket by the inflowing water from the filling line (blue line) and the magnetic stirrer. The cage surrounding the stirrer allows the eggs to be suspended but prevents contact between the flea and eggs. Eggs are then syphoned from the bucket and through two separators (red line). The separators slow the flow rate of eggs past the sensors. As the eggs pass the sensors/photovoltaic cells a signal is generated by the change in light intensity and is amplified by the amplifier. The signal is interpreted by the Powerlab SP8 and analysed by Chart 5 software. The flow rate can be altered by adjusting the height of the stand and separators. The blue line represents the filling line and the red line represents the route the eggs follow through the system.

b) Trace produced by Chart (5) software; each peak is an 'event' which indicates an egg passing the photovoltaic cells. This is created by the egg causing a change in light intensity that is detected by the photovoltaic cell. The size of the peak is related to the strength of the signal. The trace presented has been calibrated to only record signals from eggs and not signals of a lower strength. Each coloured trace represents the input from one sensor and 'events' were counted automatically using the Chart (5) software to give total egg number.

During soaking, the eggs were gently agitated to encourage separation and care was taken to remove the eggs prior to any sign of egg rupturing. Any samples that showed signs of egg rupture were discarded. Once eggs were separated, they were rinsed in clean salt-water and preserved again in a 4% formal saline solution. It was observed that to reduce the chance of egg rupture during counting, further preservation for a minimum of 72 hours was required to re-harden the eggs.

An automatic egg counter was developed following Bycroft (1986). A syphon and gravity fed arrangement was used as this greatly reduced the pressure in the system, further reducing the chance of egg rupture. The use of the magnetic stirrer in the bucket was observed to crush eggs during operation, therefore a frame was constructed around the stirrer using 1 mm mesh filter material. This allowed the stirrer to suspend the eggs within the bucket without encountering any of the eggs. The use of photovoltaic cells and the Powerlab Sp8 replaced the automatic particle counter (Figure 5.3a). The eggs breaking the emitted light beam from each cell as they passed it generated a peak as observed in Figure 5.3b. To improve precision the cells were placed in a light – tight container with one of the cells orientated horizontally and the other located vertically. Further accuracy was ensured by passing each sample through the counter twice, generating four counts. The mean of the four counts was taken to give an estimate of egg number, if there was a spurious count (greater than 20% of the mean), this was discarded, and a mean taken from three counts.

Prior to processing any samples, the egg counter was calibrated. Known samples that were counted by hand (n = 21) ranging from 100 – 6000 eggs were passed through the system to assess the error for the count. Linear regression analysis was conducted to generate the equation:

$$N = (ac \times 1.179) - 18.02 \tag{3}$$

where N represents the actual number of eggs and ac represents the number of eggs estimated by the automatic counter (Least Squared Regression, $F=1577$, $r^2 = 0.99$, $P < 0.001$).

This equation was applied to all subsequent counts from the automatic counter to calibrate total egg number (N) estimates.

5.3.2.4 Dry Weight

Egg samples were placed in a pre-weighed foil tray and excess moisture removed using paper towels. Samples were placed in an oven and critical point dried at a temperature of 100°C for a minimum of 24 hours. A sub-sample of eggs were removed from the dried egg mass. The eggs in the sub-sample were counted three times and a mean taken, the sub-sample eggs were then weighed (Lizárraga-Cubedo *et al.*, 2003). The total egg number estimated from dry weight was calculated using the equation:

$$N = td * (sn/sd) \tag{4}$$

where N represents the number of eggs, td represents the total dry weight of the egg mass, sn represents the number of eggs in the sub sample and sd represents the dry weight of the sub sample.

5.3.2.5 Developing a Proxy

The eggs removed from P2 were treated and counted in the same way as the rest of the samples (wet weighed, separated, and counted using the automatic counter and then dried and dry weighed. As it could be difficult estimating egg number on P2 from visual guides in the field, it was decided not to subject the P2 samples to the visual estimation count.

5.3.3 Biometric measurements as factors within the model

To investigate which morphological features could act as the most suitable predictors of egg number, biometric measurements as described by Conan *et al.* (2001) (excluding chelae) were taken for all ovigerous lobsters using Vernier callipers. These were the carapace length (measured from the sub-orbital spine to the posterior end of the carapace) and carapace width (measured at the widest point). The width of each abdomen segment (recoded as AW 1-5, with 1 being anterior and 5 being posterior) was also recorded for each ovigerous lobster. All measurements were recorded to the nearest 0.1 mm. Morphological parameters of the lobster (including P2) were investigated to assess if they can act as a suitable predictor for egg number. Carapace length and width and the width of each abdominal segment was recorded. Model selection based on morphological characteristics was assessed testing the null hypotheses '*there was no significant difference in the models generated using morphological characteristics to determine egg number in H. gammarus'*.

5.3.4 Data Analysis

5.3.4.1 Comparison of methods

The egg number determined by the different methods (automatic egg counter, visual, wet and dry) conformed to a normal distribution (S-W, $p > 0.05$). Therefore, least square regression analysis was applied to analyse the accuracy of each of the methods when compared to the actual number of eggs derived from the automatic egg counter. Suitability of each method was determined by assessing the accuracy of the model. The model with the highest r^2 value and the lowest Akaike Information Criteria (AIC) value was deemed as the most suitable.

5.3.4.2 Biometric measurements comparison

Carapace length is the most commonly used predictor when determining egg number in lobsters (see

Table 5.5). Each of the three models were used to predict egg number using carapace length as a predictor. The accuracy of the methods were compared (using r^2) and each model generated (using each carapace length as a predictor for each method) was compared using ANCOVA, testing the null hypotheses *'there was no significant difference in the number of eggs estimated for each of the methods when carapace length was used as a predictor for egg number'*.

Carapace length, carapace width and abdomen width (1-5) measurements conformed to a normal distribution (S-W, $p > 0.05$). Stepwise regression analysis was applied to determine the significant morphological factors that may determine egg number using the different methods. This was applied for each of the visual, wet, and dry methods trialled, for both the morphological measurements and the inclusion of a P2 proxy.

5.3.4.3 Model selection

Model selection was determined by conducting an analysis of variance (ANOVA) on the squared residuals of the most suitable models generated. Each model was compared against a progressive scale of error tolerances. Error tolerances were defined as what error in the egg number estimate a researcher could tolerate (i.e. a 5% error could be deemed as acceptable whereas a 20% error in the estimate would not be)The accuracy of each model generated was presented graphically at a progressive scale of error tolerances. All analysis was conducted using base R statistical software (R Core Team, 2017) and graphical representation using GGPLOT2 (Wickham, 2009).

5.4 Results

5.4.1 Comparison of methods

Egg number was determined using the three methods described; wet weight, dry weight and estimate visual. All samples were counted using each method (n = 45) and the wet, dry and eye methods compared to the actual count using linear regression (Figure 5.4, Table 5.1).

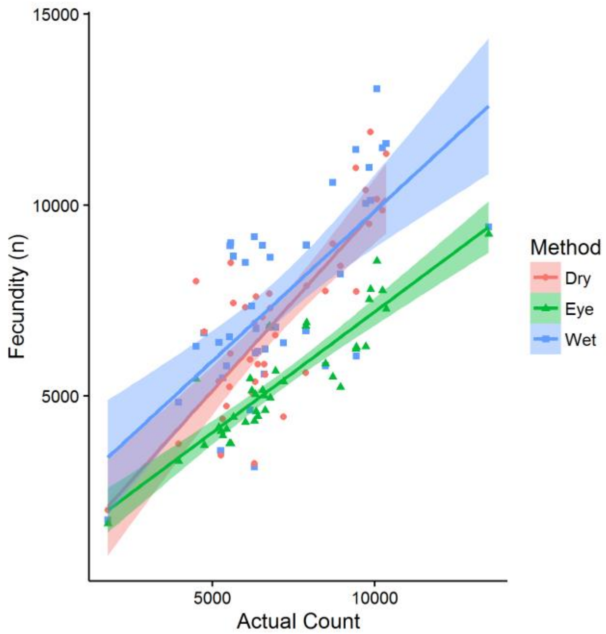


Figure 5.4: Scatterplot of fecundity estimate (egg number) determined by wet weight, dry weight and the mean of the eye estimates (y) against actual counts made using the egg-counter (extrapolated to from the calibration (x). Plotted lines represent the linear model for each method plotted. Shaded areas represent the 95% confidence intervals for each of the models.

The egg number estimate determined by linear regression for the different methods varied when actual count was used as a predictor. Egg number determined by eye was the most suitable model with the highest r^2 value, highest F value and lowest AIC value whilst egg number determined by wet weight had the lowest r^2, lowest F value and highest AIC value. Although dry weight had the closest intercept (Figure 5.4) the r^2 and F values were not as accurate as for the visual method.

Table 5.1: Results of linear modelling of differing response against actual count of egg numbers.

Response	a	a - Std. error	β	β - Std. error	R2	Model p value	F value	AIC
Wet	1998.35	961.4	0.78	0.13	0.47	<0.001	35.5	734.9
Dry	437.06	828.8	0.94	0.12	0.63	<0.001	65.6	697.6
Visual	888.34	368.2	0.63	0.05	0.81	<0.001	164.6	623.0

5.4.2 Carapace length as a predictor

To replicate the most common and widely used method of determining fecundity (Table 5.5), all three methods were analysed using CL as a predictor. Figure 5.5 demonstrates that for the different methods of egg determination, both the intercept and the slope, when using CL as predictor, did not create a robust model with high accuracy. The results of linear modelling (Table 5.2) show that if CL is the predictor used then the eye method has the highest r^2 value and had the highest level of significance for the model. However, use of this method should be taken into context of the effect size within the model, 14 – 23 observations for each sample as opposed to a single value for the other models.

When compared to actual count, egg number determined by wet weight (Figure 5.5b) presented data at the lower end of the frequency distribution whilst dry weight presented data at the upper end of the frequency distribution (Figure 5.5c). Egg number determined by eye (Figure 5.5d) had observations at all estimates that were not present when compared to actual count.

Comparison of CL against the number of eggs estimated by each of the methods (visual, wet and dry) produced a model that was not representative of the egg number of a lobster ($p > 0.05$). There were large numbers of data points falling outside the 95% confidence intervals for all methods (Figure 5.5a - d), implying that CL is a poor predictor when determining fecundity of lobsters.

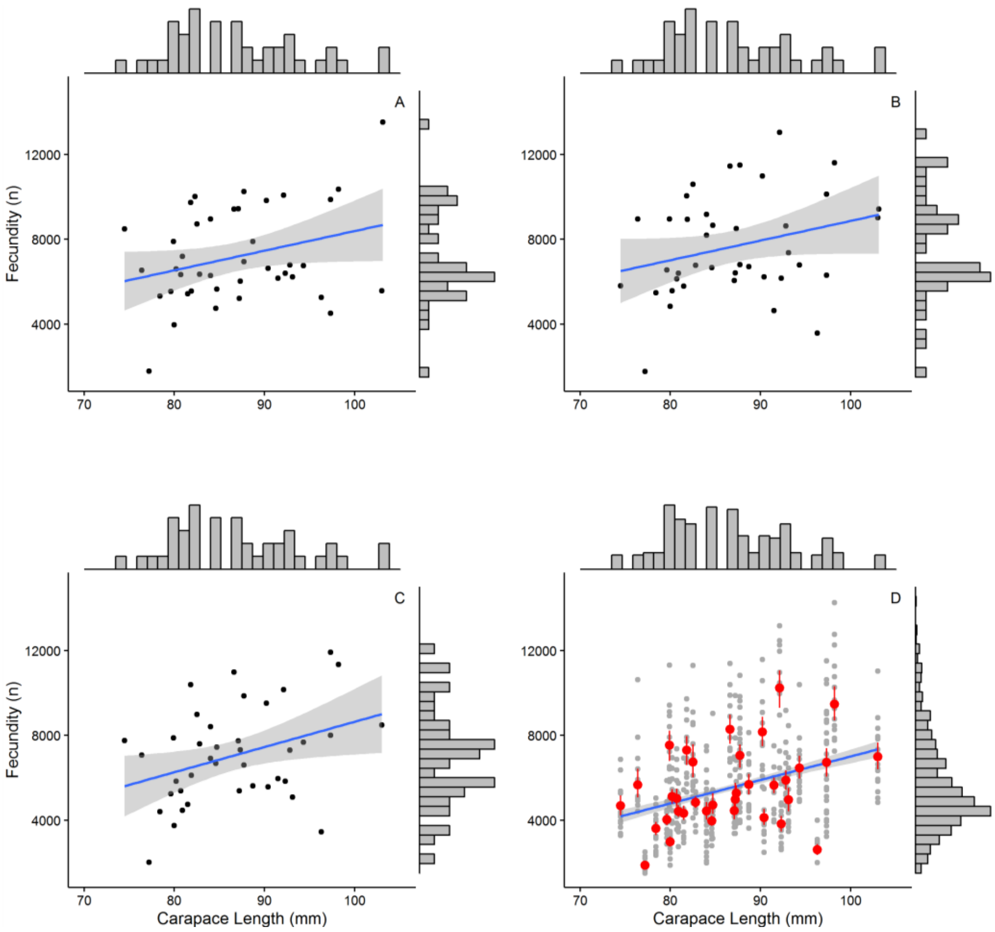


Figure 5.5: Scatter-histograms for the four different methods of estimating egg number for lobsters (y) against the carapace length (x). The four methods plotted, actual count (a), wet weight (b), dry weight (c) and eye (d). The line plotted is the regression line determined by linear modelling and the shaded area is the 95% confidence intervals of the model. The histogram on the top of the plot is the frequency distribution of the carapace length and the histogram on the right side of the plot is the frequency distribution of the egg number determined by the specific method. The red points on plot D represent the mean estimations of the eye samples (with standard deviation) and the range in individual observations plotted in grey (n = 14-23 for each eye sample).

Table 5.2: Results of linear modelling of differing methods for determining fecundity using carapace length as a predictor. The individual models were used to predict the number of eggs on a lobster with a CL of 90mm, for comparison the number of eggs predicted from the actual count is presented

Response	a	β	r^2	p- value	F value	Egg number of a 90 mm CL Lobster
Wet	-402.64	92.72	0.05	>0.05	3.202	7942
Dry	-3263.6	119.1	0.11	<0.05	5.474	7456
Eye	-3747.9	105.88	0.13	<0.05	7.54	5871
Actual Count	-879.62	92.71	0.06	>0.05	3.775	7464

5.4.3 Multiple response variable models

The results of stepwise regression analysis using the Wet, Dry and Eye methods and CL, carapace width (CW) and abdomen width (AW) 1 - 5 as the dependant variables and the actual count of egg number as the response variable produced varied accuracy of models (Table 5.3).

Table 5.3: Results of stepwise regression analysis for the three different methods trialled using all the morphological measurements recorded.

Method	**Morphological measurements within final model**	**P value**	**R2 Value**	**F statistic**	**AIC**
Wet	WW, CL, AW4, AW5	< 0.001	0.52	11.88	742.78
Dry	DW, AW5	< 0.001	0.40	14.51	750.11
Eye	Estimate by eye, CL, CW	< 0.001	0.82	59.7	667.27

The strongest stepwise model generated (referred to as morphological models) was using estimate by eye (Table 5.3) generating the model:

Egg number = (***Estimate by eye * 1.34***) + (CL * -115.56) + (CW * 162.46) + 2435.0

Using the wet weight of the egg mass generated a reasonable model (Table 5.3), however different morphological characteristics were deemed as integral to the model:

Egg number = (***WW * 108.27***) + (***CL * -169.20***) + (AW4 * 204.14) + (***AW5 * 111.20***) + 3514

The use of the dry weight method produced the poorest model (Table 5.3) but also the simplest:

Egg number = (DW * 372.67) + (AW5 * 95.04) + 81.39

The inclusion of the number of eggs on the second pair of fully formed pleopods (P2) as an additional predictor variable increased the accuracy of the models generated (Table 5.4) (referred to as P2 models). Actual count of P2 and the eye method generated the most accurate model with the highest r^2 value and lowest AIC value:

Egg number = (***Estimate by eye * 1.40***) + (***CL * -149.94***) * (CW * 227.12) * (Actual count of P2 * -0.53) + 2851.32

The wet weight method to count the number of eggs on P2 (take the overall weight, count, and weigh a sub-sample to generate egg number) produced a stronger model than using the actual count of P2 (Table 5.4):

$$\text{Egg number} = (\boldsymbol{WW * 69.79}) + (\text{AW5} * 77.41) + (\boldsymbol{P2\ estimated\ by\ WW * 1.19}) - 1147.44$$

Table 5.4: Results of stepwise regression analysis for the three-different method trialled using all the morphological measurements recorded and the number of eggs on P2.

Method	Measurements within final model	P2 measurement used	P value	r^2 Value	F	AIC
Wet	WW, CL, AW4, AW5, WW of P2	Estimate by WW of P2	< 0.001	0.54	10.69	741.24
Dry	DW, Actual Count of P2	Actual Count	< 0.001	0.48	19.79	744.05
Eye	Estimate by Eye, CL, CW, Actual Count of P2	Actual Count	< 0.001	0.83	47.27	665.51

5.4.4 Validity of the visual method

The models created using the eye method for determining egg number were more accurate than the other methods presented. However, the method should be taken in the context of the power within the model. For example, the eye model using CL as predictor did not conform to a normal distribution (Shapiro Wilkes test, $p < 0.05$) and there was a tendency for all observers to underestimate egg numbers (Figure 5.6a). The greater number of data points for the visual estimation model, collectively produced a stronger model than the other methods. However, analysis of individual estimates from the pool of observers produced models with r^2 values ranging from 0.29 - 0.81 (Figure 5.6b). The university staff (experienced field biologists) used for the estimates were more precise and had significantly higher r^2 values than the undergraduates used (Wilcox Test, W = 91, $p < 0.01$) (Figure 5.6b).

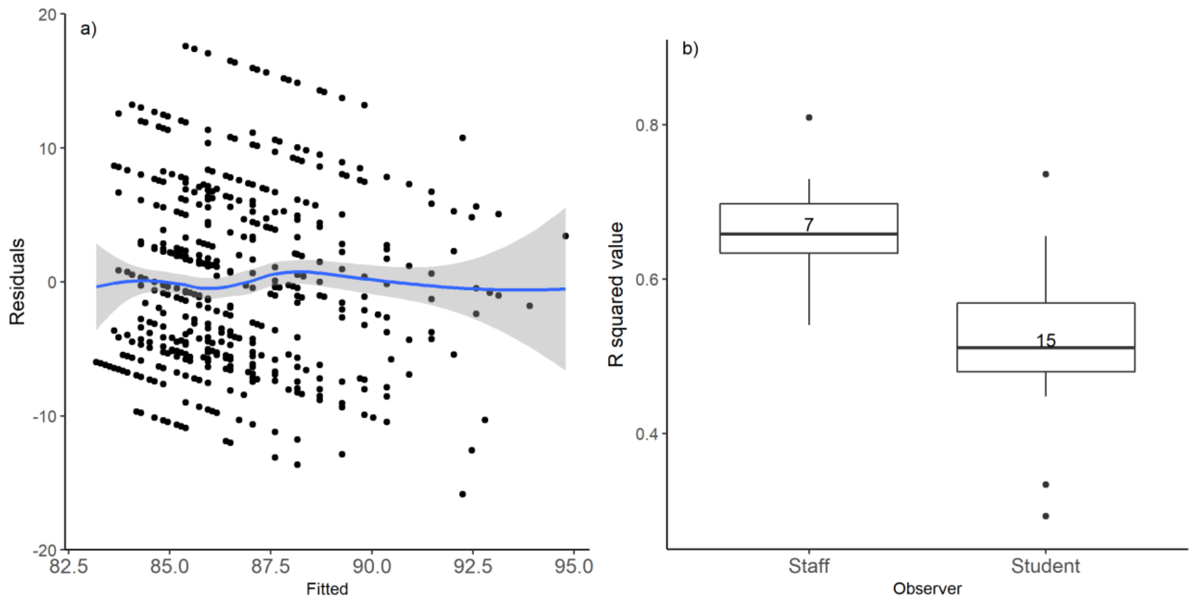


Figure 5.6: Validation plots of the eye method. a) Residuals plotted against fitted values for linear model using estimates of egg number by eye (response) and carapace length (predictor). Shaded area represents the 95% confidence intervals for the residual data. b) r^2 values determined by linear modelling of individual observers estimating egg numbers by eye. The points represent the individual r^2 value recorded. The median r^2 value is represented by the bold horizontal line; the top of the box above the median represents the 75th percentile with the bottom of the box representing the 25th percentile. The whole box represents the interquartile range (25-75), the whiskers represent the 95% confidence intervals of values of the estimates for egg number, data outside of the whisker represent outliers outside of the 95% confidence interval. This applies to all subsequent boxplots. Vales in each box represent the sample size for each factor.

5.4.5 Model Selection in relation to model error

Stepwise regression modelling improved the accuracy of the Wet method (Figure 5.7). Error tolerances were defined as what error within the egg number estimate was acceptable, these were compared on a progressive scale. At 5% error tolerance the P2 model was the most accurate (25% accuracy) in comparison to the wet method (20% accuracy) and morphological model (17.5% accurate). Accuracy of the morphological model was greatest between 10 & 20% error tolerance and beyond 20% tolerance the morphological and P2 models demonstrated a similar accuracy (< 5% difference at all error tolerances).

The Dry method showed similar accuracy of estimates between both models generated (< 5% difference at all error tolerances). However, the both the morphological and P2 models generated greater precision than the method alone (Figure 5.7b).

Both the morphological and P2 models demonstrated a much greater accuracy than the Eye method alone (> 35% more accurate up to 30% error tolerance (Figure 5.7c). The Eye method was also the least accurate when compared to the Wet and Dry methods whilst the Eye morphological and P2 models were more accurate than the same Wet and Dry models at < 20% accuracy (Figure 5.7a - c).

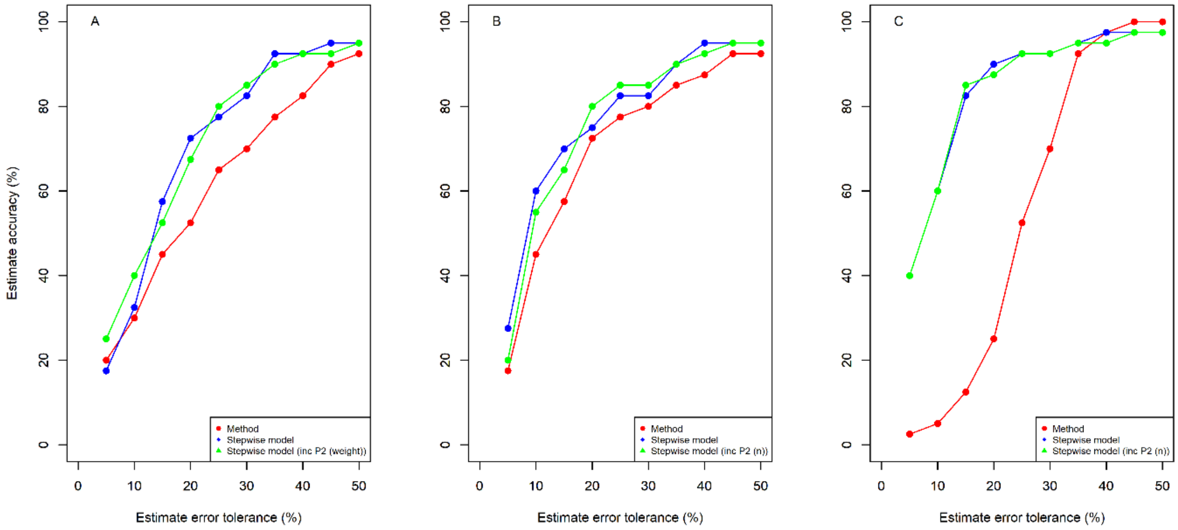


Figure 5.7: Plots representing the accuracy of each model generated (x) in relation to error tolerance for each of the models. The individual method accuracy with no morphological variables are in red. Stepwise regression models using morphological measurements are in blue and the stepwise regression models using both morphological measurements and an estimate of P2 are in green. The Wet methods a), Dry methods b) and Eye methods c) are presented.

Although the visual models were most accurate at low error tolerances (Figure 5.7c) and demonstrated a statistically stronger model, the concerns discussed bring the validity of the visual models into question. Therefore, the visual models were removed from model selection.

There was no significant difference in the squared residuals of the generated models created to predict egg number (ANCOVA, df = 3, F = 0.24, $p > 0.05$). Post Hoc analysis identified no significant difference in the generated models in all cases (Tukey, $p > 0.05$). Therefore, the null hypothesis '*there was no significant difference in the models generated using morphological characteristics to determine egg number in H. gammarus*' was accepted.

Table 5.5: Fecundity estimates and models for various methods used to determine egg number for *H. gammarus*, studies are ranked in accordance with their r^2 value. Results from Chapter 5 modelling included in **bold**.

Author	Year	Species	Fecundity estimate	Method	Model	Predictor	a	b	r^2	n	Location
Coleman *et al.*,	2019	*H. gammarus*	*potential and actual*	non-invasive measurements of volume of the ellipses of the egg mass	Power	CL	0.0107	2.962	0.95	116	Orkney Islands
Latrouite *et al.*,	1984	*H. gammarus*	*potential*	wet weight and weigh sub-samples	Linear	CL	465.87	-37021	0.9	68	Northwest France
Coleman *et al.*,	2019	*H. gammarus*	*potential and actual*	non-invasive measurements of volume of the cylinder of the egg mass	Power	CL	0.0132	3.0091	0.88	116	Orkney Islands
Agnalt *et al.*,	2008	*H. gammarus*	*potential*	wet weight whole egg mass then sub sample 1-1.5g and count eggs	Power	CL	0.0045	3.2214	0.86	215	Southeast Norway
This study	**2019**	***H. gammarus***	***potential***	**Visual estimation using visual guides in pots**	**Linear**	**CL + CW**	**1.34 + -115.56 + 162.46**	**2435**	**0.82**	**22**	**Bridlington**
This study	**2019**	***H. gammarus***	***potential***	**Visual estimation using visual guides in pots**	**Linear**	**Actual egg number**	**888.34**	**0.63**	**0.81**	**22**	**Bridlington**
Latrouite *et al.*,	1984	*H. gammarus*	*potential*	wet weight and weigh sub-samples	Linear	CL	305	-22759	0.7	70	Northwest France
Ellis *et al.*,	2015	*H. gammarus*	*potential*	wet weight whole egg mass then sub sample 2.2-3.9g and count eggs	Power	CL	0.0066	3.1	0.68	52	Cornwall
Tully *et al.*,	2001	*H. gammarus*	*actual*	oven dried (@60), weigh whole egg mass then weigh/count 3 sub samples	Power	CL	0.0044	3.1554	0.65	398	Ireland
Lizarraga-Cubedo *et al.*,	2003	*H. gammarus*	*potential*	oven dried (@100), weigh whole egg mass then weigh 4 sieved sub samples of 50 eggs	Power	CL	0.002	3.3004	64	89	Hebrides

Author	Year	Species	Fecundity estimate	Method	Model	Predictor	a	b	r^2	n	Location
This study	**2019**	***H. gammarus***	***potential***	**dry weight and weigh sub-samples**	**Linear**	**Actual egg number**	**437.6**	**0.94**	**0.63**	**40**	**Bridlington**
Hepper & Gough,	1978	*H. gammarus*	*potential*	oven dry (@100) whole egg mass, weigh mass and sub sample (weigh/count)	Linear	CL	217.74	-12490	0.62	99	West England
Roberts	1992	*H. gammarus*	*unknown*	model parameters described in Agnalt 2008	Linear	CL	400.24	-27461	0.62	53	South England
Lizarraga-Cubedo *et al.*,	2003	*H. gammarus*	*potential*	oven dried (@100), weigh whole egg mass then weigh 4 sieved sub samples of 50 eggs	Power	CL	0.0003	3.6199	0.57	31	Southwest Scotland
This study	**2019**	***H. gammarus***	***potential***	**wet weight and weigh sub-samples**	**Linear**	**WW + CL +AWF +AW5**	**108.27 + -169.2 +201.14 + 111.20**	**3514**	**0.52**	**40**	**Bridlington**
Bennet & Howard	1987	*H. gammarus*	*unknown*	model parameters described in Agnalt 2009	Linear	CL	190.1	-9629	low	83	East England
Bennet & Howard	1987	*H. gammarus*	*potential*	wet weight and weigh sub-samples	Linear	CL	427.6	-32259	high	83	Southwest England
This study	2019	*H. gammarus*	*potential*	wet weight and weigh sub-samples	Linear	Actual egg number	1998.5	0.78	0.47	40	Bridlington
This study	**2019**	***H. gammarus***	***potential***	**dry weight and weigh sub-samples**	**Linear**	**DW + AW5**	**372.6 + 95.04**	**81.39**	**0.40**	**40**	**Bridlington**
This study	**2019**	***H. gammarus***	***potential***	**Visual estimation using visual guides in pots**	**Linear**	**CL**	**-3474.9**	**105.88**	**0.13**	**22**	**Bridlington**
This study	**2019**	***H. gammarus***	***potential***	**dry weight and weigh sub-samples**	**Linear**	**CL**	**-3263.6**	**119.1**	**0.11**	**40**	**Bridlington**
This study	**2019**	***H. gammarus***	***potential***	**wet weight and weigh sub-samples**	**Linear**	**CL**	**-402.64**	**92.71**	**0.05**	**40**	**Bridlington**

5.5 Discussion

5.5.1 Comparison of methods

The two most common methods (estimates from wet and dry weight) and a visual proxy for determining egg number were compared against actual count as determined by the automatic egg counter. The visual method most closely resembled the actual count however validation of the method was discounted due to its inherent inaccuracies. The wet method was the poorest model demonstrating a low r^2 value whilst the dry method produced a reasonable model. The residual moisture retained during the processing of the eggs for the wet method could explain the unsuitability of the model when compared to the actual count, adding error to the weighing and thus egg number estimations. This extra mass has the potential to create an overestimation of the egg mass reported. Within brood variability in egg size could also play an important part in misrepresenting the number of eggs within a brood (Marshall & Keough, 2007). Eggs are extruded over a period of days (Pandian, 1970) and their size is related to their development stage (Charmantier & Mounet-Guillaume, 1992). Use of larger, more developed eggs when calculating egg number could lead to an under- representation of the egg number within a brood and vice versa with later extruded eggs. Not all eggs within a brood are viable (Moland *et al.*, 2010) and non – viable eggs may have a different mass to viable eggs (Wickins *et al.*, 1995). These factors can cause an over or under-estimate of egg number when using the wet weight method. Recording of egg size within this study may have highlighted within brood variability and further enhanced the estimations of egg number. The comparison of the different methods highlighted issues associated with each technique, however Hypothesis 5.1 was accepted as the results produced the most suitable method to be used when compared to an automatic counter.

The process of drying the eggs removes the possible inaccuracy of egg estimates due to the presence of moisture within the egg mass. The sieving of the funiculae (bonds between eggs) from the egg sample has the potential to further reduce error in the egg number estimation. (Campbell & Robinson, 1983; Lizárraga-Cubedo *et al.*, 2003). As reported, the dry weight method produced a more suitable model than that of wet weight. Of the three methods compared to the actual count, dry weight was the most suitable method used considering inherent problems associated with eye estimations.

5.5.2 Morphological characteristics as a predictor for egg number

Carapace length is the most common predictor/measurement used when determining fecundity and size in lobsters used in both the literature and industry. Lobsters have indeterminate growth (growth occurring throughout their lifetime as opposed to stopping at maturity) (Gendron & Sainte-Marie, 2006) that varies depending on factors such as nutrient availability, sea temperature and shelter availability (Wahle, 1992; Aiken and Waddy, 2009). These factors can also influence fecundity (Linnane *et al.*, 2008; van den Brink *et al.*, 2011, 2012) therefore it would be expected that there would be a strong relationship between fecundity and CL. In this study however, CL demonstrated to be an unsuitable predictor for determining fecundity in lobsters, this is in contradiction to the wider literature. For all three methods where CL was used to determine egg number, the models produced were all poor. Eye was the best method but produced a model with the r^2 value of just 0.12. Discrepancy in the estimates of egg numbers determined by each model can be seen in the histograms in Figure 5.5. The different methods produced different fecundity estimates based upon the same CL. Therefore, it appears that whatever method is used to count the egg number, in contradiction to general practice, CL is a poor morphological measurement to use to create fecundity models, therefore Hypothesis 5.2 was rejected. Addition of further variables to the model such as abdomen width or adding a sub-sample proxy (such as P2) to egg number models may increase the accuracy of the models. However, the variability in the accuracy of CL as a predictor, observed in this thesis, may be due to the accuracy of the automatic egg counter. The automatic egg counter was calibrated using a sample size of up to 6000 eggs, numbers beyond this were extrapolated from the model. This could account for the variability in CL models when compared to the wider literature.

5.5.3 Model selection

Stepwise regression analysis of all variables determined that wet weight was the most suitable method for counting eggs on lobsters, using the morphological measurements WW, CL, AW4 & AW5. However, the practicality of measuring all the variables within the model is not suitable. In addition, using the wet weight method has the potential to generate inaccuracies in estimates as previously discussed. The same method was identified when P2 was added to the model, however, both models were deemed as more suitable to the dry weight method based on the r^2 and AIC values. ANCOVA of the selected models did not demonstrate a significant difference between any of the generated models. Indicating that there is no one method that is statistically more suitable than another.

The accuracy of the models did however show different trends in relation to their error tolerances. Use of the methods: wet weight or dry weight to estimate egg number, demonstrated the lowest accuracy across all tolerances for both the wet and dry methods. Using a combination of morphological characteristics (including P2) and the wet or dry weight of the egg mass produced models with an improved accuracy in relation to error tolerances, therefore Hypothesis 5.3 was accepted. The dry weight models produced estimates that were slightly more accurate than the wet weight models. Although there was no significant difference in either of the wet or dry models generated, based on the accuracy plots generated, it is suggested using the dry weight of the egg mass and applying the equation:

$$\text{Egg number} = (DW * 372.67) + (AW5 * 95.04) + 81.39 \tag{5}$$

Although this model was not as accurate as the wet weight model, the accuracy of the estimates was the most accurate at the lower error tolerances (< 15% error tolerance). Replacement of CL with AW5 is also feasible for field scientists as it only requires a single morphological measurement to be recorded. Additionally, this method does not require an estimate of egg number from drying, weighing and counting a sub sample of eggs. The overall dry weight is used as opposed to estimating egg number (Figure 5.8).

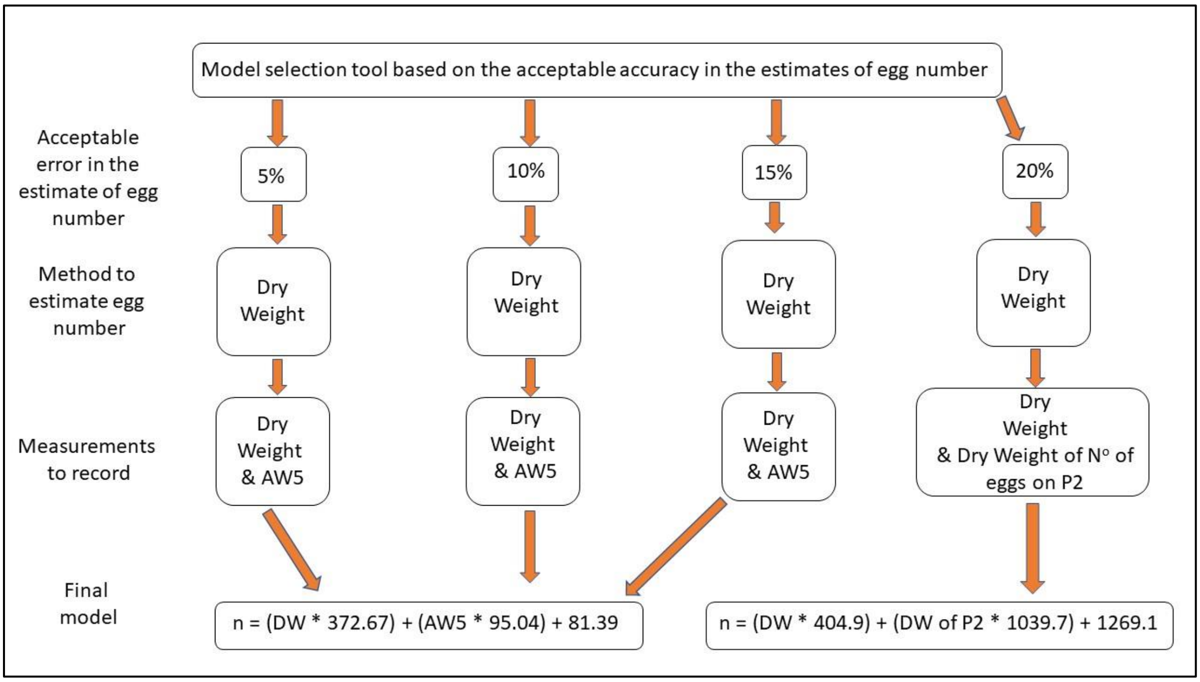


Figure 5.8: Decision schematic for the most suitable method to use to estimate egg number, measurements to record and the model selection in relation to acceptable error in the estimate of egg number.

The addition of P2 into a model can be of use in populations where removal of potential larvae for fecundity estimates could be detrimental to the stocks. The most commonly used methods for determining fecundity (apart from estimating by eye) involves removing the entire egg mass from the berried lobster (Hepper and Gough, 1974; Lizárraga-Cubedo *et al.*, 2003; Tallack, 2007a; Linnane *et al.*, 2008). In the presented studies (

Table 5.5), the number of lobsters that had eggs removed ranged between 31 and 398. This loss of eggs from the breeding stock (if collected in large quantities) could affect the viability of threatened stocks (Currie *et al.*, 2010). Removal of just the eggs on the first fully formed pleopod (P2) accounts for approximately 23% (s.d. +/- 9.9%) of the entire egg mass allowing the rest to contribute to the stock's viability. Further investigation is needed to produce an accurate proxy model using P2 considering the differing egg stages within a brood and the impact of maternal size on the proportion of eggs extruded on the second pleopod. There are 3 egg stages based on yolk components as described by Sibert *et al.*, (2004) whereas there are four stages described by Coleman *et al.*, (2019) based on visible characteristics of the eggs. Accurate proxies for egg number using the number of eggs on P2 would need to be generated for each of the egg stages. Egg stage was not recorded within this study, therefore the proxy presented should consider this.

Technological methods were not investigated in this Chapter. Image processing software (e.g. ImageJ) can be used to estimate egg number in smaller crustacean species such as *Armases cinereum* (Figueiredo *et al.*, 2008). This method is suitable for species with smaller eggs as separation is needed between eggs for the software to identify individual eggs. For use in *Homarus* spp., it may be needed to sub-sample the egg mass or process the eggs over several images.

5.5.4 Validity of the visual method

Egg estimations by eye, based on predetermined figures or as in this study, pots containing known numbers of eggs are a quick and easy method for determining egg number without having to remove the eggs from the animal. Field observations using images of the ventral side of a berried lobster as a visual guide can produce larger numbers of fecundity estimates without the need to remove the eggs (Talbot *et al.*, 1984b). The experience of the observer in estimating biological samples can affect the estimates greatly (Holden & Ellner, 2016). Direct judgement can be made with certain accuracy using empirical samples for judgement basis (Fiedler, 2000), however this may not be the case with all observers. For example, the r^2 values for individual observations ranged from 0.29 - 0.81, with only 40% of the observers demonstrating estimates with r^2 values > 0.6, of which 31% were University staff experienced in ecological estimations. The variability of the residuals within the eye model using CL as the predictor demonstrates the inaccuracy of the method. Most of the residuals lie outside of the 95% confidence intervals showing less variability with smaller samples and greater variability with larger samples. It may be possible to sub-sample larger samples for visual estimates; however, this would need to be validated.

Enumeration of large numbers is more time consuming and prone to error than smaller numbers (Trick & Pylyshyn, 1994), however visual estimations are less costly and need less resources than either of the wet or dry methods. The stage of egg development may also cause misinterpretation, observers may find it easier to estimate larger objects than smaller. Early stage eggs of recent extrusion form a denser egg mass which can be difficult to estimate due to greater occurrences of occlusion. Whereas eggs that are close to hatching do not form as dense a brood as there is estimated to be considerable egg loss during the brood cycle (Perkins, 1971; Leal *et al.*, 2013). Estimations of fewer, larger eggs should be more accurate than a larger number of smaller eggs. Although it is possible to estimate the fecundity of large numbers of females to produce large quantities of fecundity estimates using the eye method, the results can be influenced too heavily by the observers experience for the methodology to be reliable.

5.6 Conclusions

This study has shown that there are varied issues associated with each of the methods tested for determining egg number in lobsters. The accuracy required of the egg number estimates may determine which method is used. For example, when estimating the number of eggs on a 90 mm CL lobster, there was only a difference of ~ 500 eggs between the actual count and the wet and dry methods; however, the estimate from the eye method was over 1500 eggs lower than the actual count. Therefore, the eye estimation is not recommended as a suitable method for determining egg number in lobsters. Carapace length has also been shown to be an inaccurate predictor when modelling egg number in lobsters. The number of eggs on the second pleopod has been demonstrated to be a suitable contributor when combined with other morphological measurements. Although there were no statistical differences between the generated models, the accuracy of the egg number estimate varied between models. Adoption of the models by the scientific and fisheries management community is feasible, however, the selection of the best method and variables to record is dependent upon researchers' resources. The findings of this chapter can help researchers and managers in adopting suitable practices for choosing the best method for determining fecundity in lobster estimates to fit their resources. The implications of which can further aid in their correct management and assessment, striving towards sustainability of their stocks.

Chapter 6: Does the closure of an area due to wind farm construction improve the egg yield of the ovigerous lobster population in the site?

6.1 Abstract

Closed areas can be used by fisheries managers to protect spawning and nursery grounds of commercially important stocks. To date, their use, however, has been to protect spawning finfish species as opposed to mobile invertebrates. The implications of a closed area due to wind farm construction was investigated on the density, size structure and egg production of a population of ovigerous lobsters and the implications of subsequently reopening the site. A basic model suggests that closure can have a positive effect on the size and density of ovigerous lobsters. Egg yield of individual lobsters did not differ significantly in the wind farm between the open and closed period. However, total egg yield was greater in the wind farm during the closed period, due to the greater proportion of larger female lobsters observed in the wind farm during the closure. Post opening of the site, there was a shift towards smaller female lobsters, however density of ovigerous females and egg yield matched the control site. It is suggested that the closure of a site can have a positive effect on total lobster egg yield, however this needs to be taken in the context of recent legislation changes and a sufficient duration of the closure being suitable to detect long-term effects.

6.2 Introduction

Spawning stock biomass is generally assumed to be one of the governing factors that controls the subsequent stock recruitment into a fishery (Planque and Fredou, 1999; Stockhausen *et al.*, 2000; Zhang *et al.*, 2011) although Szuwalski *et al.*, (2015) suggests environmental factors affect the stock recruitment greater than its spawning stock biomass. Stock managers often have a suite of tools they can use to protect the spawning stock biomass. These can include quota and total allowable catch limits (European Parliament, 2019), technical measures such as mesh size of trawls, length of tow (Vogel *et al.*, 2017), engine size, days at sea restrictions, closure of nursery/spawning grounds and seasonal closures (Murawski *et al.*, 2000; Hart, 2009; Cohen and Foale, 2013b; Cohen *et al.*, 2013b). For example, gadoid fisheries are managed using most of these measures to ensure sustainability (Köster *et al.*, 2014)

In crustacean pot fisheries there are fewer management tools available. In the UK, stock management of crustacean fisheries targeted via pots has been through a minimum landing size (MLS) approach. This is a process by which a fisher is prohibited to retain an animal below their stated MLS. In UK lobster fisheries, this has increased to match the increased effort associated with lobster fisheries to its current size of 87 mm carapace length (CL), although Scotland has increased its MLS to 88 mm CL aiming to increase to 90 mm CL by 2019 (Scottish Government, 2019). Sundelöf *et al.* (2015) observed that MLS was not sufficient to protect the spawning stock in lobster populations. Recent national legislation passed on October 1st, 2017, prohibited the landing and retention on deck of ovigerous (subsequently referred to as berried) *H. gammarus* and *Palinurus sp*. in UK waters (MMO, 2017). Prior to this, berried lobsters were permitted to be landed providing they were greater than the proscribed MLS. This measure has been met with scepticism by many in the industry as a broad-brush approach by managers without listening to measures put forward by industry, such as a national increase in the MLS (Mike Cohen, pers. comm. former CEO, HFIG). There is also concern that the measure will shift fisheries focus, at certain times of year, to the male population only. Sex ratios of male to female lobsters in the Holderness fishery are approximately 1.1:1 (Roach, 2019). Size and sex specific fishing mortality of male lobsters have been linked to fisheries inducement towards smaller body sizes due to mating behaviour changes (Sørdalen *et al.*, 2018). Current stock assessment of the male lobsters in the Holderness (Yorkshire and Humber stock unit) (Cefas, 2017) is deemed as heavily exploited. The increased pressure on the male population, due to the berried ban, may prevent the male population reaching Maximum Sustainable Yield (MSY) targets and require more stringent management measures in the future. In contrast, in the State of Maine (USA), *H. americanus* fisheries, have been subjected to a berried ban since 1872 (Kelley, 1992; Acheson, 1997). This protection of berried females has been attributed (amongst other factors) to the scale and success of the *H. americanus* fishery, reaching 162,547 tonnes of *H. americanus* landed in 2017 (FAO, 2019). There are additional management measures used in *H. americanus* fisheries. For example, the Gulf of Maine fishery enforces for commercial fisheries: night time closures during summer months, minimum and maximum landing sizes, bait restrictions, storage at sea restrictions, berried ban and mandatory 'V' notching (see Chapter 2, Figure 2.7), escape vents in all pots and biodegradable escape panels, pot design, pot limitations and mitigation for whale entanglement (Kelley, 1992; State of Maine Department of Marine Resources, 2009).

The benefit of protecting berried crustaceans can be observed in the *Cancer pagurus* fishery in England, where protection of berried *C. pagurus* has been in force since 1877 (Bennett, 1995). However, there are likely other factors contributing to large *C. pagurus* populations such as habitat types and high fecundity levels in comparison to *H. gammarus* (several million eggs produced by *C. pagurus* compared to thousands by *H. gammarus*) (Tully *et al.*, 2001; Haig *et al.*, 2016). In 2019, landings into the UK of *C. pagurus* were 34,036 tonnes (£76.5 million) and *H. gammarus* 3248 tonnes (£42.5 million), demonstrating a higher abundance of *C. pagurus* in UK waters (MMO, 2019a). *H. gammarus* retains a higher market value than *C. pagurus* (average of £10.84/kg greater in 2019 (MMO, 2019a)), however the value in *C. pagurus* in in landing higher quantities. It is still legal to land berried *Nephrops norvegicus* in UK, however most landings are from mobile gear (76.7%of landings by value in 2015 (Russell and Solutions, 2017) where targeting is less selective than pot fisheries (Johnson and Johnson, 2013). The benefits of protecting berried lobsters, may not be observed for some years due to recruitment into the fishery taking approximately 5 – 8 years (Sheehy *et al.*, 1999).

The combination of closed areas and rotational/seasonal harvesting strategies can protect the spawning stock biomass of target species. However this can be more effective for finfish (specifically reef fish) than invertebrates, due to invertebrates being more susceptible to intense fishing pressures once a site is opened (Cohen and Alexander, 2013; Cohen *et al.*, 2013b, 2013a). Short or seasonal closures may not be beneficial to slow growing species and have been shown to be ineffective when closed for less than 10 years (Gnanalingam and Hepburn, 2015). The Norwegian *H. gammarus* fishery observed a 90% reduction in their *H. gammarus* stock between 1960 and 1980 indicating stock collapse due to over-exploitation (Moland *et al.*, 2013b). A series of small scale (< 1 km^2) Marine protected areas (MPA) were created to protect the *H. gammarus* population (Pettersen *et al.*, 2009). Within the MPAs, after 4 years, individual size of lobsters and CPUE was increased (Moland *et al.*, 2013b). This matches previous studies reporting increases in CPUE and biomass of lobsters within MPAs/NTZs (Davies *et al.*, 2015; Watson *et al.*, 2016; Howarth *et al.*, 2017). Spill-over effects have also been observed in lobsters from closed areas. Thorbjørnsen *et al.* (2018) and Howarth *et al.* (2017) observed spill-over of larger lobsters from Skagerrak MPAs and Lamlash Bay marine reserve respectively. Huserbråten *et al.,* (2013) observed limited overspill (4.7%), and gene flow was reported from within an MPA due to larval dispersal from the site. However, Watson *et al.*, (2016) reported a genetically homogenous population that was not different to within the Lundy Island NTZ. The increased population of larger lobsters within a closed area may not have benefits with regards to egg production. Koopman *et al.,* (2015) observed that brood size of larger lobsters was lower than expected, although this could be linked to a 31% decline in fecundity within the observed population which was attributed to increased temperature regimes in the study area. Accurate fecundity estimates for lobsters are essential in generating yield per recruit models which form part of the stock assessment and potentially their application to closed areas.

Fecundity (which is a realized measure of fertility) of a lobster can be classified into two categories or fertility terms: potential and actual fecundity. Potential fecundity is the maximum number of eggs a female can carry (normally immediately following extrusion) whereas actual fecundity is the number of eggs that may produce viable larvae at the end of the brood period (Stechey and Somers, 2008). Actual fecundity can be affected by factors such as temperature, inter/intra-species competition, abrasion during movement and handling of lobsters caught via commercial fisheries during the brood period (Mori *et al.*, 2001; Wright, 2013; Green *et al.*, 2014; Koopman *et al.*, 2015; Coleman *et al.*, 2019). However, these factors are often in question as to their effects. For example: Ellis *et al.*, (2015) observed no correlation between fecundity and mean temperature and Goldstein and Watson III (2019) observed a reduced brood period in lobsters where the eggs were subjected to a constant temperature as opposed to seasonal fluctuations indicating temperature has low impact on fecundity in lobsters. However Koopman *et al.*, (2015) attributed a decline in fecundity to a temperature shift in the study area.

Not all eggs extruded are fertilised, whole clutches and portions of clutches have been observed to be unfertilised (Johnson *et al.*, 2011). Although legislation is now protecting berried lobsters from being landed, actual fecundity may still be affected as the interaction between commercial fisheries and egg bearing females still exists as berried lobsters are still caught prior to release. The use of closed fishing areas may be a tool that fisheries managers can use to enhance the protection of berried lobsters and thus generate a more robust stock.

6.1.1 Aims, hypotheses and objectives

The aim of Chapter 6 is to understand the effects of closing an area due to offshore wind farm construction, to fishing exploitation, has on the density of ovigerous lobsters and the egg yield of lobsters in the site.

The hypotheses to be tested in Chapter 6 were:

6.1. The density and size structure of ovigerous lobsters observed in 2015 will be positively affected by the closure of the Westermost Rough offshore wind farm development to fishing exploitation during the construction phase when compared to the control site and subsequent post-construction period.

6.2. The individual and total egg yield from ovigerous lobsters observed in 2015 will be positively affected by the closure of the Westermost Rough offshore wind farm development to fishing exploitation during the construction phase when compared to the control site and subsequent post-construction period.

The objectives of Chapter 6 were:

- An estimate of effective trapping of a lobster are will be calculated from the literature in order to estimate the density of ovigerous lobsters at the study sites.
- Comparisons of density and size structure of ovigerous lobsters will be made between the wind farm and the control site during the closed period and also during the open period. Interactions between sites and the open and closed period will also be investigated.
- Estimates of egg yield for individual lobsters and will be generated based on the results of Chapter 5. Comparisons will be made of individual egg yield of lobsters between the wind farm and the control site during the closed period and also during the open period. Interactions between sites and the open and closed period will also be investigated. Total egg yield for each site will be generated and compared for both the closed and open period.

6.2 Methods

At sea sampling was conducted in accordance with the protocols described in section 2.3.2. Female lobsters that were sampled at the Westermost Rough offshore wind farm site during the 2015 survey (July – September 2015) were used for the analysis conducted in this chapter. The OWF site had been closed to fishing exploitation for a 20-month period between January 2014 and August 2015. The control site was not closed to fishing throughout this period. The OWF site was reopened to fishing exploitation on the 13th August 2015. The 2015 data were separated into two periods, the closed period (3rd July – 15th August 2013) and the open period (15th August – 27th September 2015). To account for seasonal fluctuations, the control site was also separated into the closed and open periods for comparison to the wind farm site (Figure 6.1).

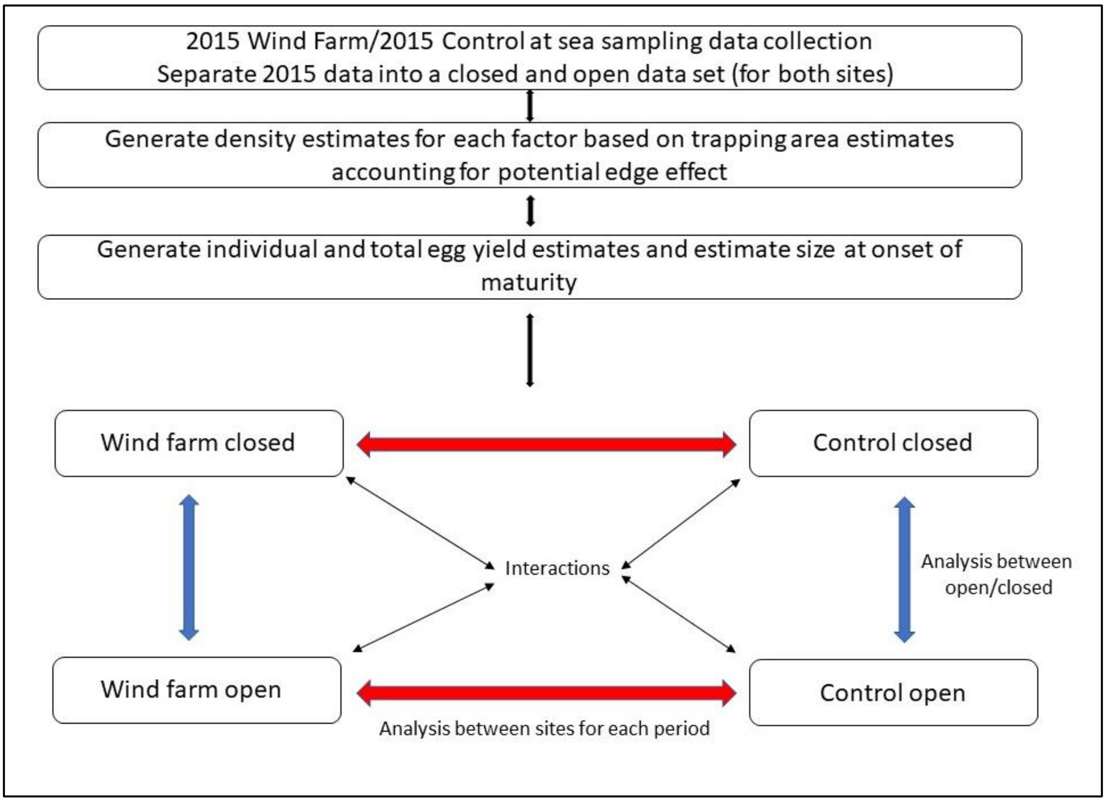


Figure 6.1: Schematic of experimental design for Chapter 6.

To be able to assess egg yield from the berried lobsters, density of female lobsters sampled at the different sites/periods was needed. Estimation of population density from catch statistics can be difficult without a quantifiable area sampled. In fish sampling, area of sampling net, time towed and water flow rate can be used to estimate the volume of habitat sampled (Bell *et al.*, 2001). With pot fisheries this is not possible although Bell *et al.* (2001) presented three areas of interest with regards to estimating fishing area of a pot: 1. area of influence, the area where the bait can be detected and alters the behaviour of the target species. 2. trapping area, the area where the probability of catching the target species is greater than zero, and 3. effective area fished, a notional area containing all the animals that were sampled, the effectiveness of this area reduces with distance from the pot. These areas can be influenced by many factors such as immersion time of the pot, water current direction and speed, proximity of pots in the string and bait degradation. Bell *et al.* (2001) also proposed the "edge effect" of a string of lobster pots. Whereby the end pots in a string have a greater catchability than the other pots in the string. An estimate of whether edge effect is present is needed in order to estimate population density. This is due to the area of influence of each pot within the string, overlapping the area of influence of the pot next to it. The combined scent from two or more adjacent pots may also be more powerful than the end pots which may confuse the lobsters searching for the bait, the end pots may be easier to locate. The pots at the end of the string only have overlap on one side thus reducing the conflicting influence of another pot. To assess if the sample strings demonstrated edge effect and whether overlap was present, a non-parametric Kruskall Wallis was used to test the null hypotheses '*there was no significant difference in median number of lobsters captured between the 30 pots located in the survey strings'*.

There was a significant difference in the median number of lobsters caught between pots within the strings deployed in the wind farm (K-W, Chi Sq. = 56.894, df = 29, $p < 0.01$). However: post Hoc analysis (Wilcox, pairwise comparison) did not highlight a significant difference in lobster abundance among individual pots ($p > 0.05$ in all cases). There was no difference in median lobster abundance caught between pots in the strings deployed in the control site (K-W, Chi Sq. = 28.34, df = 29, $p > 0.05$) (Figure 6.2). Therefore, the null hypothesis '*there was no significant difference in median number of lobsters captured between the 30 pots located in the survey strings'* was accepted indicating edge effect was not observed.

Pot 30 demonstrated slight variability to other pots within the string in both sites, demonstrating a higher median value than the immediate pots close to it (3 pots away). However, pot 1 did not follow the same trend (Figure 6.2). Whilst pot 30 was a fine mesh pot (Figure 2.3), the amount of bait used was the same as the other post, therefore it was assumed that the catchability/attractiveness of the fine mesh pots was the same as the coarse mesh pots.

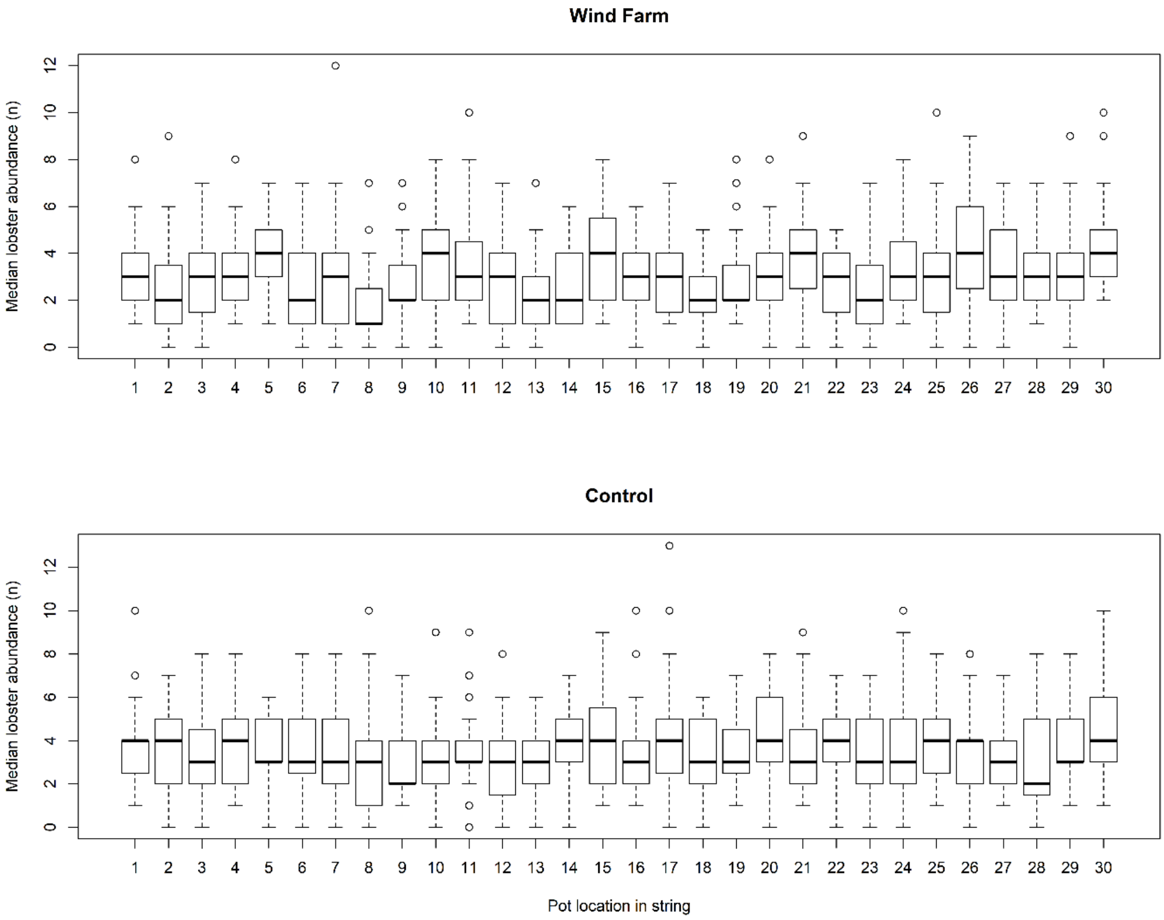


Figure 6.2: Boxplots of median abundance of lobsters sampled in each of the pots located within the strings (n = 23) in both the wind farm and control sites of the WMR OWF.

Moland *et al.* (2011) estimated a home range of *H. gammarus* between 5728 – 45,548 m^2. Watson *et al.* (2009) calculated a trapping area of 2604 m^2 (a circle with a radius of 28.8 m) for *H. americanus.* This included the area of bait influence and the home range of the lobster. The addition of trapping area as well as home range made for a more accurate assessment of trapping area. In the absence of estimations of trapping area for *H. gammarus*, it was assumed that the 2604 m^2 effective trapping area reported by Watson *et al.* (2009) can be applied to *H. gammarus* fisheries. The trapping area estimate used in this chapter does not account for the factors affecting pot attractiveness discussed in previous chapters (3 & 4) such as; inter/intra species competition, bait attractiveness, alternate food source availability and other factors such as water current and the presence of other fishing gear in the area. For the trapping area estimates these factors were assumed to be constant.

In order to obtain an accurate estimate of lobster density within the two sites, trapping area of the entire string was estimated for both the assumption that there was no pot overlap and an estimation when pot overlap was present. Trapping area for both estimates used the Watson *et al.*, (2009) estimate of 2604 m^2. For all estimates, it was assumed the fine mesh pots had the same trapping area as the larger parlour pots due to the fact the same amount of bait was used in all pots.

To estimate trapping area of a string without overlap, it must be assumed that the trapping areas are ≤0.5/distance between the pots. In this case it was assumed that the radius of the trapping area was equidistant between the pots to ensure maximum trapping area was accounted for. Each pot was 36.576 m apart, the radii of the trapping area without overlap was 18.288 m giving a trapping area of each pot of 1050.71 m^2 and a total trapping area of a string without overlap of 31,521.26 m^2.

To estimate trapping area of a string with overlap, each pot within the string was given a trapping area with a radius of 28.8 m determined by Watson *et al.*, (2009). The total trapping area of 30 pots without the presence of overlap was 78,172.88 m^2. The pots within the string were 36.576 m apart, this generated an overlap of 20.024 m. The area of overlap of the pots was calculated using SketchUp Pro 2018 (Version 18.0.16975). Half of the area of overlap (324.87 m^2) within the string was removed from the area estimate of a string of 30 pots to account for the influence of one pot next to another in a string (Figure 6.3). The total trapping area when accounting for overlap (and thus edge effect) was 68751.65 m^2.

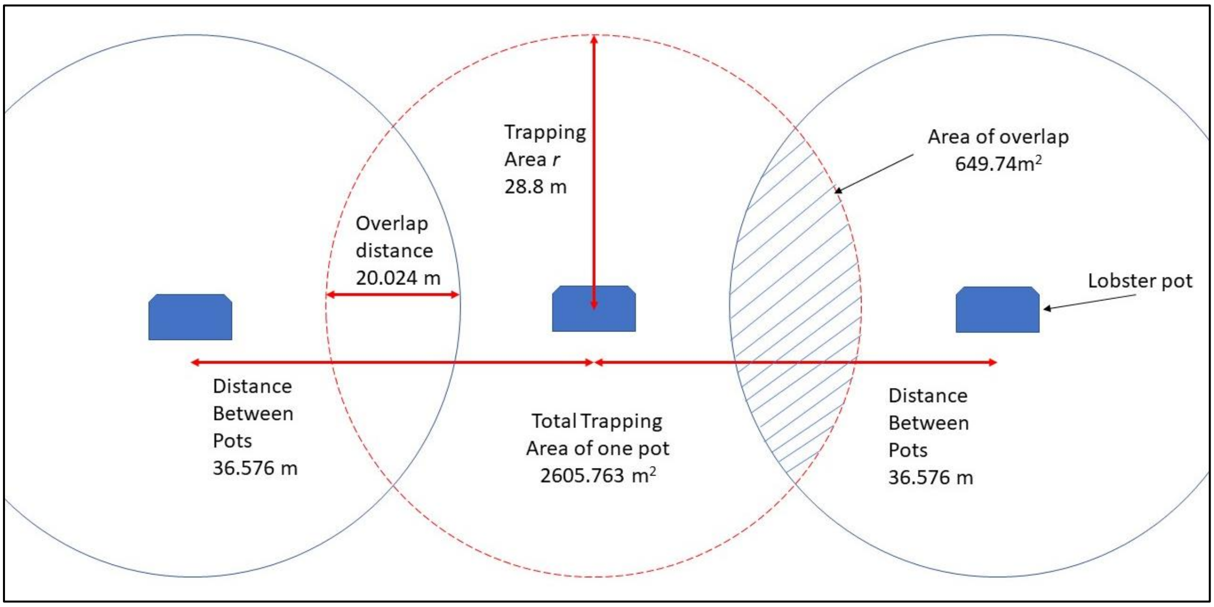


Figure 6.3: Schematic of trapping area overlap in relation to a string of pots using the Watson *et al.*, (2009) estimate.

6.2.1 Data analysis

All analysis was conducted using R statistical software (R Core Team, 2017). Packages ggplot2 (Wickham, 2009), gridExtra (Auguie, 2017), dplyr (Wickham *et al.,* 2017), car (Fox and Weisberg, 2011), Matching (Sekhon, 2011) were used for data manipulation, analysis and graphical outputs.

6.2.1.1 Density of ovigerous lobsters

Density estimates for berried lobsters was generated for the wind farm and control sites and the open and closed period based on the trapping area estimates generated above. Estimates of the berried lobster density (mean number of berried lobsters/total trapping area) within each site was calculated for both trapping areas estimates and scaled to *n* berried lobsters/100 m^2. A 20% retention rate was also applied to the density estimates using the formulae:

$$\text{Density} = \frac{\left(\frac{100}{T-area}\right)*abun}{retention\ rates} \tag{6}$$

where *T-area* equates to the trapping area estimated and *abun* equates to the abundance of berried lobsters sampled in the string.

Density of berried lobsters conformed to a normal distribution (S-W, $p > 0.05$ in all cases) and the variances could be assumed to be equal (Levene's test, $p > 0.05$) in all cases. A two-way ANOVA was applied to the density of berried lobsters for both area estimates (no overlap and overlap assumed), to test the null hypotheses *'there was no significant difference in the density of ovigerous lobsters between the wind farm and control site during the closed period and also between the wind farm and control site once the wind farm had been reopened to fishing exploitation (open period) and no significant interaction between site and closure period)*'.

6.2.1.2 Size distribution and size at onset of maturity

The berried lobster size data (raw and Log transformed) did not conform to a normal distribution (S-W, $p <0.05$ in all cases). Therefore, two-sample K-S test was applied to the size data to analyse the differences in length frequency distribution of berried lobsters between the wind farm and control sites and between the open and closed period. Size at onset of maturity (SOM) is determined at the point where 50% of the females in a population are berried (Tully *et al.*, 2001; Linnane *et al.*, 2008). Assessing SOM using functional indicators is the most common method when estimating maturity in lobsters, via the direct observation of eggs being carried by a female (Wood, 2018). SOM estimates were calculated by plotting the proportion of female lobsters that were observed to be berried against the CL and visually assessing the size at which 50% of the females in the population were berried.

6.2.1.3 Egg production

For fecundity estimates, potential fecundity (maximum yield) was used to calculate egg yield due to variables needed for actual fecundity estimates not being measured. CL was shown to be a poor predictor for fecundity in Chapter 5. However, egg number and egg stage were not recorded during the Westermost Rough OWF survey. CL was the only morphological variable recorded for each berried lobster. In the absence of other morphological measurements recorded in the WMR OWF study, it was necessary to select one of the models using CL as a predictor (Figure 5.5,Table 5.2).

It was necessary to select a model where the size of the berried lobster (determined by CL) was the predictor for egg yield. Whilst the visual method produced a model with greater accuracy, it was discarded due to questions about the validity of the method being raised (Section 5.4.4). The model generated using CL and the 'wet' method did not produce a significant model, therefore the model using CL as a predictor and the 'dry' method (Table 5.2) was used to determine individual egg yield for each size of berried lobster:

$$N = (CL * 119.1) - 3263.6 \qquad (7)$$

Where N = the estimated egg number and CL is the Carapace length of the berried lobster.

Egg yield for each of the sites and closure periods was calculated by estimating the egg number for each size of female lobster sampled and the proportion of those lobsters that were berried, using the equation:

$$Yield = \ nEggs\ Size * nlobsF * pBerried \qquad (8)$$

Where nEggs Size is the number of eggs estimated for each size based on CL of lobster, nlobsF is the number of female lobsters in the population and pBerried is the proportion of the female population that is berried. This model assumes 100% of lobsters that are caught are retained in the pots.

The retention rates (the catchability of the gear type) of lobsters caught using pots is unknown in UK fisheries. There is some comparison of retention rates between pot types (parlour vs inkwell pots) (Lovewell and Addison, 1991) but retention rates of UK modern pots are not currently studied. In *H. americanus* fisheries, retention rates have been estimated between 6% (with escape gaps) and 33% without gaps (Jury *et al.*, 2001) and (Karnofsky and Price (1989) observed a 24% retention of lobsters entering pots. Due to the uncertainties of UK pot retention rates and the range of pot retention reported for *H. americanus* fisheries, a conservative estimate of 20% retention rate was applied to the egg yield model:

$$Yield = \frac{nEggs\ Size * nlobsF * pBerried}{retention\ rate} \qquad (9)$$

The egg yield estimates (generated using Equation 6) for both the assumption of 100% retention and 20% retention did not conform to a normal distribution, therefore a non-parametric Wilcoxon Rank Sum test was applied to the null hypotheses '*individual egg yield of berried lobsters did not differ significantly between the wind farm and control site during both the closed and open period*'. Total egg yield for all berried lobsters was estimated in the wind farm and control site during the closed and open period.

6.3 Results

During the 2015 survey a total of 2105 female lobsters were sampled of which 262 were berried (12.5%). The greatest number of berried lobsters was observed in the wind farm area during the closure (n = 89, mean CL = 86.5 mm, s.d. +/- 10.8 mm), followed by the control site (n = 64, mean CL = 78.4 mm, s.d. +/- 7.9 mm) and wind farm (n = 59, mean CL = 79.8 mm, s.d. +/- 9.4 mm)) during the open period with the fewest observed in the control site during the closure period (n = 50, mean CL = 79.1 mm, s.d. +/- 8.7 mm)). The smallest berried lobster was 71 mm CL and the largest was 108 mm CL. Berried lobsters below the MLS of 87 mm CL accounted for 42.4% of all berried lobsters sampled and 57.6% of berried lobsters accounting for recruits into the fishery.

Of the 23 survey days conducted in 2015, 12 were during the closure period of the wind farm and 11 were sampled once the wind farm was opened to fishing exploitation. There were no missed survey days due to weather and the mean soak time was 3.9 days (sd +/- 2.1).

6.3.1 Density of berried lobsters

There was no significant difference in the mean density of berried lobsters between sites or between closure periods, however, there was a significant interaction between site and the open and closed period (status) (ANOVA, Table 6.3). Post hoc analysis identified a significant difference (Tukey, $p < 0.05$) between the mean density of berried lobsters in the wind farm (mean = 0.11, s.d. +/- 0.04) and control sites (mean = 0.06, s.d. +/- 0.03) during the closure period when overlap was not assumed (Figure 6.4a) and between the wind farm (mean = 0.05, s.d. +/- 0.02) and control sites (mean = 0.03, s.d. +/- 0.01) during the same period when overlap was assumed (Figure 6.4b).

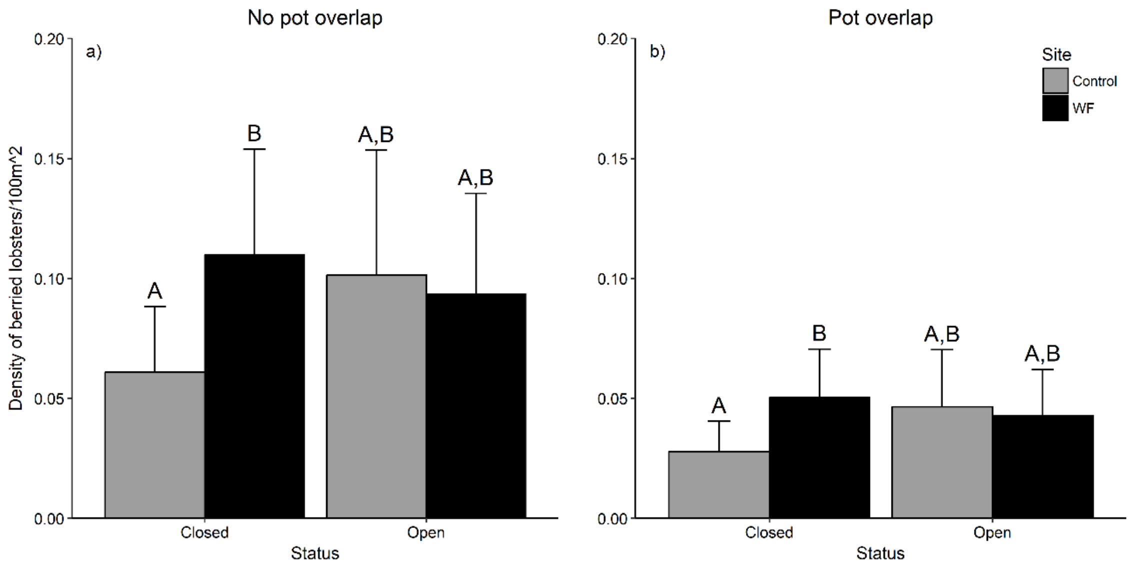


Figure 6.4: Bar plots of density of berried lobsters/100 m^2 for both trapping area estimates (no overlap (a) and overlap assumed (b)) estimated between sites and closure periods. Both estimates assume a 20% retention rate of caught lobsters. The top of the bars represents the mean value of y and the top of the error bars represent the standard deviation of y.

Therefore the null hypothesis *'there was no significant difference in the density of ovigerous lobsters between wind farm and control site once the wind farm had been reopened to fishing exploitation (open period) and no significant interaction between site and closure period)*' was accepted. However, the null hypothesis '*there was no significant difference in the density of ovigerous lobsters between the wind farm and control site during the closed period'* was rejected.

Table 6.1: Results of two-way ANOVA to determine the effects of site and status on berried lobster density. Significant factors are in **bold**.

Source	Df	Mean Sq	F	*p*
Site	1	0.007	3.885	n.s.
Status	1	0.002	0.966	n.s.
Site*Status	**1**	**0.009**	**5.275**	**< 0.05**
Error	42	0.002		

6.3.2 Size distribution and size at onset of maturity

There was no significant difference in the size distribution of berried lobsters sampled between the wind farm and controls sites either during the closed or open period (K-S, $p > 0.05$, Table 6.2). There was a significant difference in the size distribution of berried lobsters in the wind farm site between the closed and open periods (K-S, $p < 0.05$, Table 6.2). There was a shift towards smaller female berried lobsters sampled in the wind farm between periods.

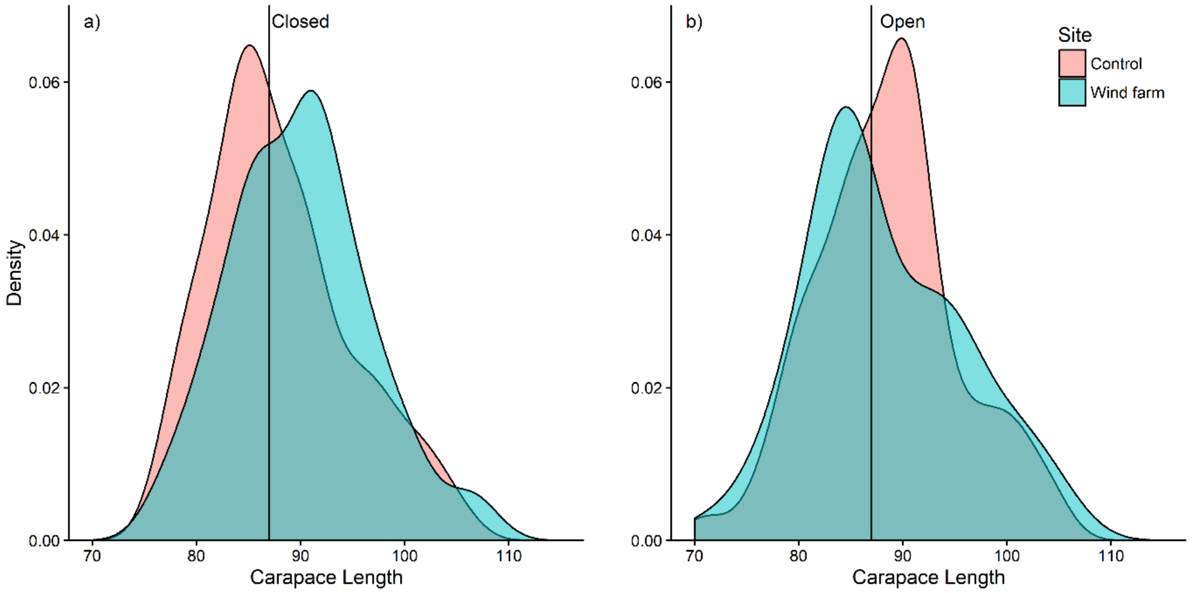


Figure 6.5: Density plots of size distribution of berried lobsters sampled between the wind farm and control sites at the WMR OWF when the wind farm during the closed period (a) and open period (b). The vertical line represents the minimum landing size of 87 mm carapace length.

The peak of the berried lobster distribution during the closure was above the MLS of 87 mm CL. After opening of the wind farm site to fishing exploitation, the peak of the distribution dropped below the MLS (Figure 6.5a & b). Size distribution of berried lobsters was not significantly different at the control site between closure periods.

Table 6.2: Results of two-sample Kolmogorov Smirnov tests, analysing the size distribution of berried lobsters sampled between the wind farm and control sites at the WMR OWF between closure periods. Significant tests are in bold.

Variables tested	Test statistic	*p* value
Wind farm and control during closure	0.2	n.s. (0.1528)
Wind farm and control when open	0.1465	n.s. (0.5256)
Wind farm between open/closed	**0.2301**	**< 0.05 (0.04588)**
Control between open/closed	0.1312	n.s. (0.7189

Visual SOM estimates at all sites demonstrated a long flat slope until approximately 80 mm CL. At this point the trend for the control site for both the closed and open period and the wind farm during the closed period follow a similar trend (Figure 6.6a, c & d). This is reflected in the visual SOM estimates that only vary between these sites and closures by 1 mm CL. The greatest variation in trend of the proportion of lobsters that were berried was observed in the wind farm site once it had been opened to fishing exploitation (Figure 6.6b). This was reflected by a visual SOM estimate that was 87 mm CL; 9- 10 mm less than the other sites and of a size that reflects the MLS in the fishery.

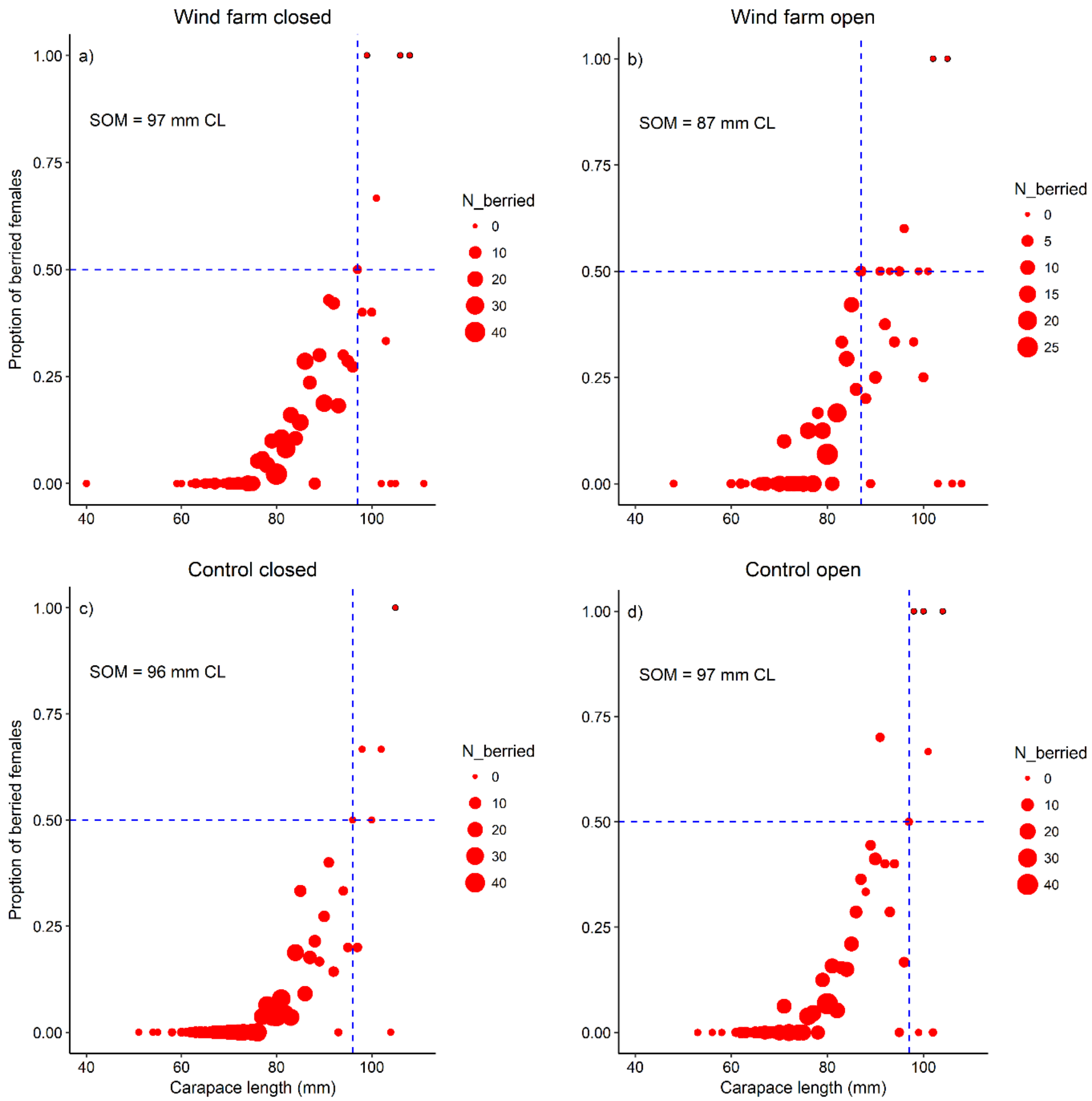


Figure 6.6: Proportion of berried female lobsters against their carapace length for the a) wind farm and c) control when the wind farm was closed to fishing exploitation and the b) wind farm and d) control site was the wind farm was open to fishing exploitation. The size of the points reflects the number of berried lobsters for that size category. The horizontal blue dashed line is the point at which the proportion of berried females is at 50%, this reflects the size at onset of maturity for the sampled population. The vertical blue dashed line indicates the size of berried lobsters that intercepts with the 50% line. SOM for each sample site and closure period is labelled on each plot.

6.3.3 Egg production

The greatest estimated number of eggs produced by an individual size class was observed in the wind farm during the closure period ($4.99 * 10^4$ (100% retention), $43.6 * 10^5$ (20% retention) eggs in the 86 mm CL size class) (Figure 6.7 a & b). The lowest number of eggs produced by an individual lobster was observed in the wind farm once the site was open to fishing exploitation ($3.91 * 10^3$ (100% retention), ($1.9 * 10^5$ (100% retention).) (Figure 6.7a & b). Median egg yield (Table 6.3) of the berried lobsters demonstrated no significant difference in the egg number produced between the wind farm and closure sites and their respective closure periods (Wilcoxon Rank Sum test, $p > 0.05$, Table 6.4).

Table 6.3: Median and range off egg yield from berried lobsters sampled between the wind farm and control sites between closure periods. Egg number was derived from model using carapace length as a predictor and the 'dry' method (Table 5.2).

Site	Closure period	Retention rate	Median yield	Range
Wind farm	Closed	100%	7,461	5,483 – 49,850
Control	Closed	100%	5,722	5,483 – 49,850
Wind farm	Open	100%	7,461	3,906 – 31,772
Control	Open	100%	5,722	4,811 – 30,699
Wind farm	Closed	20%	3,404,923	438,275 - 43,618750
Control	Closed	20%	2,499963	204,250 - 26,289120
Wind farm	Open	20%	3,404923	195,318 – 15,091780
Control	Open	20%	2,499863	360,860 – 19,263030

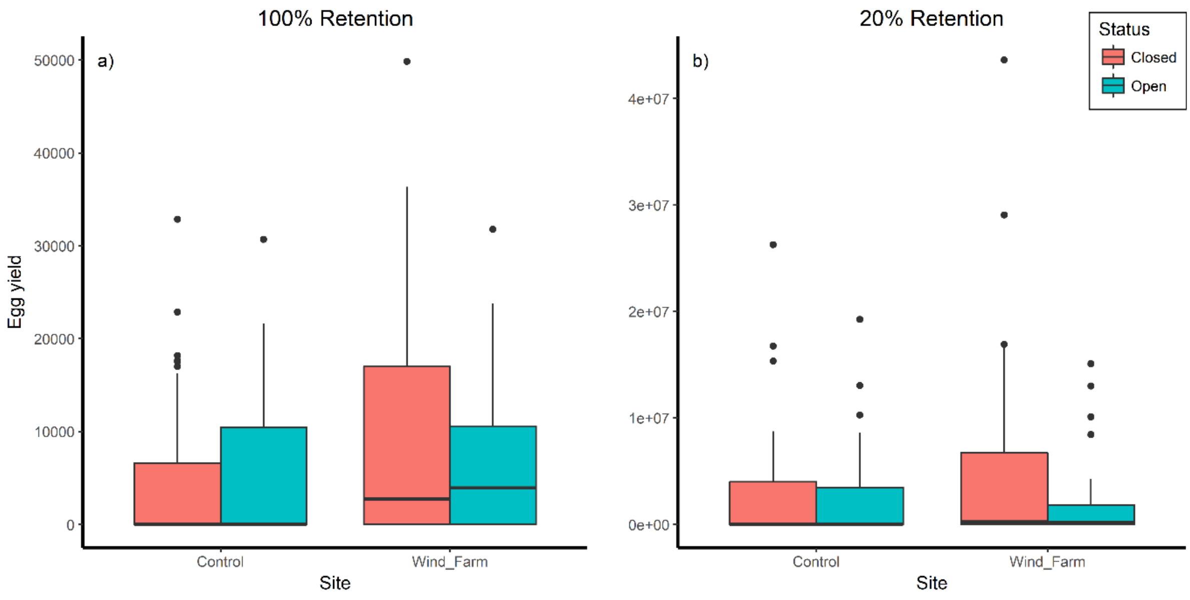


Figure 6.7: Boxplot of median egg yield of berried lobsters sampled between the wind farm and control sites and closure periods. Egg yield estimates plotted for a) 100% retention rate and b) 20 % retention rates. Note the difference in scales of the y axis. Egg numbers were generated from the model generated using carapace length as a predictor and the 'dry' method as described in Table 5.2.

The wind farm during the closure period had the greatest estimated total egg yield (n = 4.55 * 10^5 (100%), 245.9 * 10 ^{5} (20%)), 43% greater than the control site for the same period (n = 2.59 * 10^5 (100%), 127.6 * 10^5 (20%)). The opposite was observed in the wind farm site during the open period (n = 2.54 * 10^5 (100%), 78.1 * 10^5 (20%)), 7.1% fewer total egg yield produced than the control site for the same period (n = 2.73 * 10^5 (100%), 104.9 * 10^5 (20%)).

Table 6.4: Results of Wilcoxon Rank Sum tests analysing the egg yield from the berried lobsters sampled between the wind farm and control sites at the WMR OWF between closure periods. Egg numbers were generated using the 'wet' model generated in Chapter 5 (Equation 6).

Variables tested	Retention rate	Test statistic	*p* value
Wind farm and control during closure	100%	1062	n.s.
Wind farm and control when open	100%	1039.5	n.s.
Wind farm and control during closure	20%	1068	n.s.
Wind farm and control when open	20%	1074.5	n.s.

Therefore, the null hypothesis '*individual egg yield of berried lobsters did not differ significantly between the wind farm and control site during both the closed and open period'* was accepted.

6.4 Discussion

6.4.1 Density of berried lobsters

The closure of the wind farm site produced the greatest density of berried lobsters on the ground. This was significantly greater than the density of berried lobsters in the control site during the closure, therefore hypothesis 6.1 was accepted. Increased population density can increase intra-species interactions, increasing mating opportunities. However as density moves towards saturation, moulting (and potential mating during the moult) can decrease competition interactions for resources (Aiken and Waddy, 2009). Density-dependant factors determining carrying capacity of a habitat can affect population dynamics (Fogarty and Gendron, 2004). For example, shelter can be a limiting factor to mating success in lobster populations (Steneck, 2006). The potential increased habitat in the wind farm (scour stone addition) may support a greater mating population and thus the density of berried lobsters in the wind farm. Merder *et al.,* (2019) observed a density dependant suppression of growth in a *N. norvegicus* population, the greater density of lobsters may have affected the growth of the berried lobster population. Due to the limited closure duration of the site this is unlikely as this would require several moult cycles to be observed.

One of the potential benefits of a closed area is the potential for overspill of commercially exploited resources from the area into the surrounding ecosystem (Causon and Gill, 2018). Potential overspill of an increased population within a closed area has been demonstrated in the Skagerrak MPA networks (Thorbjørnsen *et al.* 2018) and *Jasus edwardsii* have been observed to be displaced from a reserve as density increased (Freeman *et al.*, 2009). In this study, density of berried lobsters has not demonstrated overspill or displacement into the control site. The mating process of lobsters and subsequent mate guarding prevents female motility until such a time as the shell re-hardens (Debuse *et al.* 1999b). The reduced density of berried lobsters observed in the control site during the closure could be due to the increased density observed in the wind farm and potential mate interactions, effecting the movement of sexually mature females between sites.

The berried lobsters observed in the control site were generally smaller. The smaller berried lobsters in the control site may have been deterred from entering the pot due to pot interaction factors thus lowering the density estimate (Addison, 1995). However, this was not observed during the open period, where the density of berried lobsters was similar at both sites. It is likely that the absence and re-introduction of fishing effort was the governing factor effecting berried lobster density within the sites. In the absence of a change in fishing effort, it is likely that the density of berried lobsters between the wind farm and control site would be similar.

6.4.2 Size distribution

At the time of the survey, fishing for berried lobsters of legal size (> 87 mm CL) was permitted within the study area, the ban on landing ovigerous lobsters was introduced in October 2017 (DEFRA, 2017). The increase in berried lobster size in the wind farm site during the closure period was expected due to the exclusion of fishing effort within the site. The control site was subjected to fishing exploitation throughout the closure period, therefore berried lobsters of legal size may have been caught by the commercial fishery. Berried lobsters below the MLS were observed in greater frequency in the control site during the closure period, however this difference was not significant. Due to the size of lobsters being a factor when determining dominance (Sheehy and Bannister, 2002; Skog, 2009), the greater frequency of lobsters of larger sizes in the wind farm site may have displaced the smaller lobsters from the wind farm site. Additionally, if a larger lobster enters a pot first they will deter other lobsters from entering. Addison (1995) observed that pots with pre-stocked large lobsters had a reduced catch rate of lobsters over a standardised immersion period. The greater amount of smaller lobsters in the control site could demonstrate overspill from a closed wind farm site. Bertelsen and Hornbeck (2009) observed that smaller *Panulirus argus* were more highly motile than their larger counterparts, although this did not equate to larger migration distances. Overspill of lobsters from marine reserves has been observed to be that of larger lobsters not smaller (Howarth *et al.*, 2017; Thorbjørnsen *et al.*, 2018b).

Lobsters migrate inshore (Smith *et al.*, 1999) to breed during the warmer months, the greatest abundance of berried lobsters has been observed inshore (inside 8 nm) in July and August (HFIG, unpublished data). Female lobsters undergo ecdysis prior to mating, requiring a soft shell for the process (Phillips, 2007) although mating can occur between moults if required (Waddy and Aiken, 1990). Post mating the male lobster protects the soft-shelled female within a shelter. This is a process called ‘mate guarding’, where the male guards the female to ensure paternity of the brood (Debuse *et al.*, 1999a; Watson *et al.*, 2017). Mate guarding takes place until the female shell re-hardens, this can last between 7 – 12 days (Debuse *et al.*, 1999b). Male lobsters generally need to be larger than the females to mate as they need to manipulate the female during mating (Phillips, 2007). The presence of larger lobsters reported in Section 4.4.1 in the wind farm during the closed period, may be ‘mate guarding’ the smaller lobsters within the wind farm, reducing the presence of smaller berried lobsters observed during the survey. Additionally, the presence of larger lobsters previously reported may be deterring the smaller berried lobsters from the pot and subsequent capture. In addition, Howarth *et al.*, (2017) reported nearly double the amount of berried lobsters within a marine reserve than observed outside.

The results here support those reported by Howarth *et al.* (2017) in the Lamlash Bay marine reserve, where a greater proportion of larger lobsters were sampled. Therefore, it is likely that the absence of fishing exploitation in the wind farm during the closure period was responsible for the differences in the size distribution of berried lobsters.

The period following the opening of the wind farm site saw a shift to smaller sizes of berried lobsters observed, this shift produced a significantly different pattern to the distribution during the closed. There was no increased frequency of larger berried lobsters observed during the closure period was absent once fishing exploitation was re-introduced. As berried lobsters could then be targeted by the commercial fishery, it is likely this reduction was due to the re-introduction of fishing effort. This indicates that the introduction of fishing exploitation to the wind farm site after a closure period, influenced the size distribution in the wind farm. However, the size distribution of berried lobsters came to reflect the distribution of berried lobsters at the control site within the period of the survey.

Inference from the SOM plots indicate that the re-introduction of fishing effort had an effect on the SOM of female lobsters in the wind farm site. During the closure period the SOM of female lobsters in the wind farm was similar to the control site (+/- 1 mm CL difference) and the same as the control site during the open period. In the wind farm site once fishing effort had been re-introduced, the SOM of female lobsters decreased to 87 mm CL from 97 mm CL. The MLS of lobsters is set at a size that ensures that most lobsters can breed at least once prior to being recruited into the fishery. The SOM estimates observed in this study, are greater than the MLS enforced within the fishery (with the exception of the wind farm during the open period). Previous studies of SOM within the fishery using offshore sampling (Wood, 2018) has estimated SOM to be ~ 92 – 108 mm CL, however this was for both sexes. Lizárraga-Cubedo *et al.*, (2003) observed a difference of 31 mm CL in the SOM of female lobsters observed between two geographically distinct lobster populations; however, this may have been influenced by a truncated size range of lobsters sampled. The same has been observed in different crustacean fisheries (Linnane *et al.*, 2008; Haig *et al.*, 2016). Actual (visual assessment of gonad development), as opposed to functional maturity of *H. gammarus* was assessed by Tully *et al.*, (2001), their SOM estimates ranged between 92.5 – 96 mm CL. The SOM estimates within this study reflect those generally found in the literature for *H. gammarus*. However, density effects in catch and wait fisheries have been linked to density dependent suppression of growth in a *Nephrops norvegicus* fishery which could affect the SOM of the target species (Merder *et al.*, 2019).The wind farm site during the closure demonstrating similar SOM estimates as the controls site indicates that the reduction in SOM once the site was re-opened was not due to the geographical differences in sites. The re-introduction of fishing effort, due to the short-term intensive effort, removed many females in the population that were greater than the MLS. This removal of the upper end of the size spectrum shifted the SOM estimate left, generating a lower estimate than the other sites/closure periods.

6.4.3 Egg production

The potential fecundity of a lobster is linked to maternal size, in which the larger the lobster the greater the surface area under the abdomen allowing for egg attachment (Moland *et al.*, 2010). Irrespective of the method used to calculate egg number in Chapter 5, there was generally a positive relationship between body size (carapace length) and the number of eggs produced (Figure 5.4) and including abdomen width variables improved the accuracy of most models. Lobsters in closed areas have been observed to be larger in size than the surrounding areas (Hoskin *et al.*, 2011; Wootton *et al.*, 2012; Davies *et al.*, 2015; Howarth *et al.*, 2017). This was also observed for the size of berried lobsters in the wind farm during the closure period. It was expected that there would be a greater median number of eggs produced in the wind farm during the closure compared to the control site, but this was not observed. Although the greatest egg yield was observed in the wind farm during the closure ($4.99 * 10^4$ (100% retention, $43.6 * 10^5$ (20% retention) in the 86 mm CL size class)), there was no significant difference in the egg yield of individual berried lobsters between the two sites, therefore Hypothesis 6.2 was rejected. The greatest abundance of berried lobsters was in the wind farm during the closure. This increased abundance generated a total egg yield of more than 40% (100% retention) and more than 70% (20% retention) greater yield than any other site or closure period.

Closure of the wind farm site resulted in an increase in the total number of eggs generated by the lobsters during this period. The brood period of *H. gammarus* is generally 9-12 months, but can be subject to temperature regimes (Branford, 1978; Goldstein and Watson III, 2019). Although there was an increase in potential fecundity observed during the closure, this may not equate to actual fecundity. The berried lobsters recorded may have been carrying eggs at varying stages of development from recently extruded to pre-release, and egg stage was not recorded during the survey. The use of closed areas to protect egg bearing females needs to account for the brood period and migration patterns of the species the closure is targeting (Moland *et al.*, 2011a, 2011b) .

The wind farm site was closed for 20 months, however, this survey was only conducted in the summer of 2015. The berried lobsters recorded could have extruded eggs at any period during the closure or immigrated into the site during the seasonal offshore/inshore migration. To accurately assess total egg yield from a closed area, data would need to be collected throughout the entire brood period of the target species and throughout the closure period of the site, ensuring egg development stage is accounted for.

The larvae from larger female lobsters are generally more viable than larvae from smaller lobsters (Leal *et al.*, 2013). This is due to larger lobsters being able to invest more lipids into the eggs during development, producing larger, fitter larvae. Lobsters have four planktonic larval stages. In the first three stages, they are passively dispersed as their motility is determined by water currents. The final stage, they can swim towards food sources and against prevailing currents before metamorphosing and settling on the seabed and adopting a benthic lifestyle (Rötzer and Haug, 2015). The larger larvae have greater feeding capacity during the planktonic phase, than smaller larvae due to their size, optimizing food sources (Mente *et al.*, 2001). The closure of the wind farm protected larger berried females from fishing exploitation. The increased total egg yield from the wind farm site was attributed to the increased abundance of larger females. Therefore, the increased egg yield due to the closure was generally from the larger female population, producing potentially the fittest larvae. Potential fecundity from the wind farm site may have contributed potentially double the quantity of larvae to the surrounding population in comparison to the control site. The recent change to national legislation, protecting all berried female lobsters may change the effectiveness of a closed area with regards to overall egg yield of the site. There will be a greater abundance of berried lobsters within the fishery, possibly increasing potential fecundity to levels similar to observed in the wind farm site during the closure (up to 70% greater total egg yield in comparison to the control during the same period). However, the increased interactions with the commercial fishery and handling of the increased number of berried lobsters may influence actual fecundity. Additionally, the increased density of berried lobsters may alter the mating strategies in the fishery or affect the rate at which maturity is reached (Merder *et al.*, 2019).

Stock assessments need estimates of the reproductive capacity of a fishery such as size at onset of maturity and yield per recruit (Caddy, 1986; Sundelöf *et al.*, 2015). Haig *et al.* (2016) observed there was no standardised methodology for determining size at the onset of maturity for *C. pagurus* populations. The same can be observed with fecundity estimations for lobster populations (discussed in Chapter 5). A standardised methodology for fecundity estimates would make comparisons of populations more effective. Lizárraga-Cubedo *et al.*, (2003) observed that whilst spatially close, two distinct

H. gammarus populations had different fecundity estimates. The egg yield estimates should be taken into context with the model used to generate egg numbers for each size class. The accuracy of fecundity estimates using just CL as a predictor are not as accurate as when combined with other morphological features or the number of eggs on the second pleopod (P2) (Chapter 5, Tables 5.1 – 5.4). Accurate estimations of fecundity for the Holderness fishery, using the appropriate methodology, may produce different results with regards to the closed area hypotheses.

6.5 Conclusions

Closure of the wind farm has demonstrated a positive effect on individual female size, egg yield and density of the berried lobster population. The absence of fishing pressure during the closure period was the governing factor during this period. Individual egg yield within the closed area, whilst not significantly different resulted in a greater total egg yield which was nearly double that of the control site, highlighting the effectiveness of a closed fishing area on total egg yield and potentially larval production. Closure of a site needs to consider the brood period of the lobster and their seasonal migration patterns, optimising periods to protect the berried lobsters. Closed areas designated to protect berried lobsters may be affected by the change in legislation since this study was conducted. Re-opening of the wind farm demonstrated a shift of the wind farm berried lobster population to reflect that of the control site.

The results of the egg yield and density estimates should be taken in the context of the assumptions made for the estimates. Accurate trapping area estimates may highlight differences in the density of berried lobsters not observed within this study. A standardised methodology for fecundity estimates could aid in more accurate estimations and comparisons of potential fecundity between areas.

Chapter 7: Discussion, suggestions for further work and conclusions

7.1 Discussion

The expansion of offshore wind developments in recent years has seen increased spatial interactions with many marine sectors such as: commercial and recreational fisheries, shipping and navigation, oil and gas industry, aviation, and the increase in designated marine conservation sites. Their developments also need to consider the environmental effects on the archaeology, geology, hydrology and aesthetics of the development. Their global expansion can be seen as a way for many European Union member states to meet their requirements of the Renewable Energy Directive (2009/28/EC), to generate 20% of a country's energy from renewable sources by 2020 (European Commission, 2016). There is public and political pressure for many countries to invest in renewable energy, offshore wind developments are often used to meet these requirements. Placing the structures offshore, many out of sight of land, offsets the "not in my back yard" issue with many terrestrial structures. One of the challenges of wind developments being located "out of sight" offshore is getting the energy generated to the grid for consumption. This often requires large-scale investment and infrastructure to facilitate energy transfer to shore. This can be via the use of extensive export cables and boosting stations to ensure maximum energy transfer. The number of boosting stations is dependent on the distance from array to shore. For example, Hornsea Four wind farm (65 km from the Holderness coast) is currently proposing three to four boosting stations to facilitate energy transfer (Ørsted, 2019b). This may increase the spatial extent of the development and its interaction with other marine users. This thesis has focussed primarily on an offshore wind development located no more than 13 km from shore and its effects on a commercially important crustacean fishery.

Chapter 1 focussed on reviewing the existing literature on lobster and crustacean fisheries and their management, closed areas as a management tool and the development of the offshore wind sector and their interaction with commercial fisheries. The importance of crustacean fisheries both on a local/regional and global scale, and their different management regimes were summarised. It was highlighted that, to date, there were few empirical studies associated with offshore wind developments and their effects on the marine environment. Literature highlighted a need for co-existence/co-location between the two industries and the requirement for more empirical studies on the ecological effects of offshore wind developments. Chapter 2 introduced the fishery that was the focus for the thesis, its history, development and current state and management regimes. The chapter also described the at-sea sampling methodology and fisheries gear used for the data collection. Chapters 3, 4 and 6 present the data collated during the at sea sampling programme and Chapter 5 presents laboratory experiments. Chapter 3 presents the results of a 3-year study investigating the effects of the WMR wind farm development on the local lobster population and also the commercial and none-commercial bycatch sampled. Chapter 4 investigated the effects of a closure of the wind farm site on the commercial and none-commercial fisheries in the area and the effects of subsequently reopening the site to fishing exploitation. In order to investigate egg yield of lobsters during the closure of the site, a methodology was needed to determine accurate fecundity estimates for lobsters, this was the focus of Chapter 5. Chapter 6 used the models generated in Chapter 5 to assess if egg yield of lobsters was increased in the wind farm site during the closure period.

7.2 Summary of principal findings

During the three-year study of the Westermost Rough offshore wind farm development (2013 pre-construction survey to the 2015 and 2017 post construction surveys), effects on the lobster population and associated commercial bycatch were observed (Figure 7.1). Lobster size structure and catch rates increased in the wind farm during the 2015 post construction survey when compared with the 2013 pre-construction survey, this was also greater when compared to the control site. However, these differences were not evident when the 2017 post construction survey was compared to the 2013 pre-construction survey. The governing factor for these differences was the closure of the wind farm site to fishing exploitation for safety reasons due to the construction phase of the development (Jan 2014 – Aug 2015).

The closure of the site, and the fact the 2015 survey straddled a portion of the closure period (3rd July – 15th August 2015) and post-opening of the site (15th August – 27th September) allowed an investigation into the closed area effects of the wind farm site and the effects of subsequent reopening of the site to fishing exploitation. In the wind farm site, during the closure period (acting as a closed area), lobster size increased although catch per unit of effort did not increase. However, the landings per unit of effort (what a fisher would land to market) was greater in the wind farm site during the closed period, indicating a population of larger, good quality (inferred from higher LPUE) lobsters compared to the control site. This was to be expected due to the control site being subjected to fishing exploitation whereas fishing mortality was not observed in the wind farm site.

The closure of the wind farm site allowed for a greater density of ovigerous lobsters, thus increasing the total egg yield from the site during the closure period. However, egg yield of individual ovigerous lobsters did not differ between the wind farm and control site for the closed and open period. The absence of fishing mortality on the lobsters in the wind farm site allowed for a greater abundance of larger lobsters in the population, increasing density of ovigerous lobsters, thus increasing total egg yield from the area.

Subsequent reopening of the wind farm site (15th August 2015) saw an immediate increase in fishing effort in the wind farm site (landings into Bridlington doubled on the 17th August 2015 in comparison to the 14th August). This intensive effort subsequently targeted those previously protected lobsters, resulting in a reduction in the size structure, catch rates and total egg yield from the wind farm site in comparison to the control site and the closed period. However, these levels did not reduce to below that of the control site, indicating that although reintroduction of effort had a short term effect, the size structure, catch rates and egg yield of lobsters reflected that of the control site within the time period of the survey in 2015 (Figure 7.1).

Commercial and non-commercial bycatch demonstrated changes to their size structure, catch rates, diversity, and community assemblages over the three survey years. The greatest change was in edible crab, size and catch rates were reduced in comparison in the 2015 and 2017 surveys compared to the 2013 pre-construction survey. However, the effects reported appeared at both the wind farm and control site for each year, indicating natural variability between the survey years as opposed to a wind farm effect. The closure effect of the wind farm site on the commercial bycatch species was compounded by the greater abundance of larger lobsters in the site during the closure, influencing catch rates of commercial bycatch species.

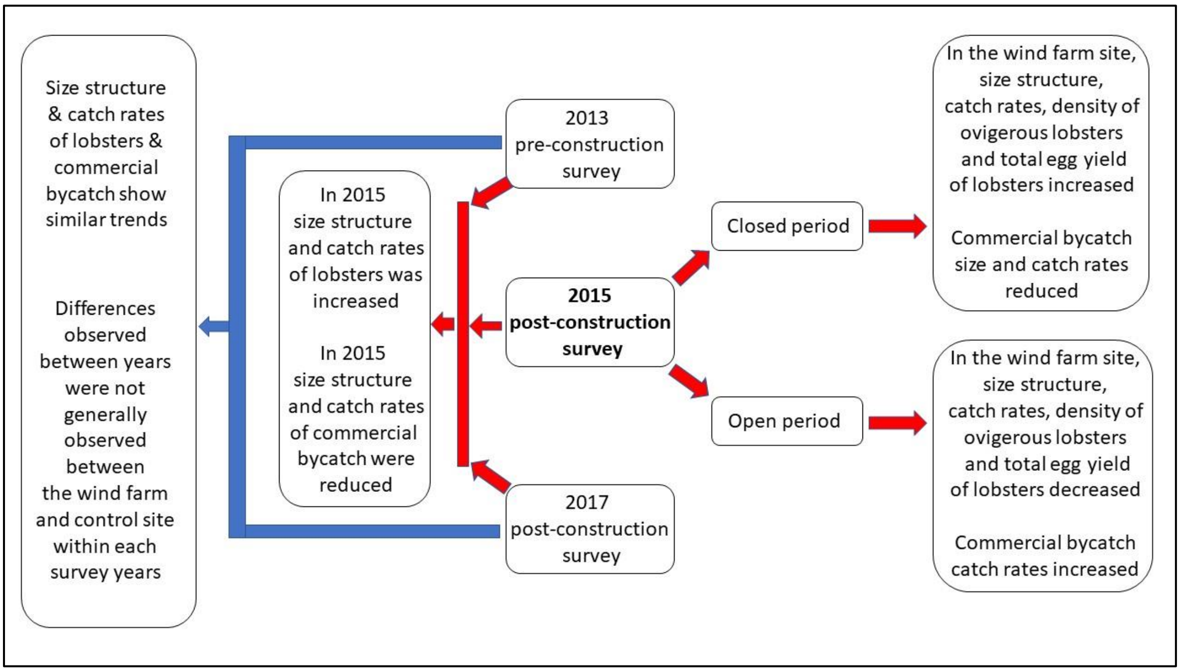


Figure 7.1: Schematic of the principle findings of the thesis. Similarities are denoted by blue arrows and differences by red arrows. The 2015 data were generally different to the 2013 and 2017 data.

7.3 Offshore wind developments and commercial fisheries

7.3.1 The expansion of offshore wind on a global scale

As the offshore wind sector expands globally, there are some technological advances being made to lessen their environmental impacts. The capacity of individual turbines has increased from 8.3 MW for the Blyth wind farm (first in the UK), to 12 MW proposed for the Dogger Bank wind farm (edf Renewables, 2019; SSE Renewables & equinor, 2019), requiring less turbines to be located offshore to generate the same amount of power. For example, the Humber Gateway wind farm located on the east coast of the UK, (12 km south of the study area) required 73 turbines to generate 219 MW of power whereas the Westermost Rough wind farm only required 35 turbines to produce 210 MW of power (Crown Estates, 2019). However, both wind farms maintain similar spatial footprints, although the spacing between turbines is greater in the Westermost Rough wind farm. The increased spacing allows for many forms of fishing to take place within the wind farm boundaries, although the area is solely used for potting. Some offshore developers consult the fishing industry during their planning stages on the best layout of the wind farm to ensure that fishing practices can take place. For example, the Westermost Rough wind farm has the turbine grid orientated to allow for the north/east, south/west orientation of fishing gear deployed in the area as opposed to the optimal layout for energy production (Hywel Roberts pers. comm. Lead Environment and Consents Specialist, Ørsted). The proposed Fécamp wind farm off the Normandy coast is considering both turbine orientation and cable routes to be orientated to allow for mobile fishing gear deployment in the area (edf, 2019).

Allowing fishing to take place at offshore wind farm sites is an initiative taken by many developers to allow for co-location of the two industries which in turn can aid in spatial conflict resolution (Christie *et al.*, 2014; Hooper and Austen, 2014b; Hooper *et al.*, 2015). Altering fishing practices in an area due to restrictions recommended for the construction phase had a positive effect on the size, catch rates and total egg yield of lobsters (reported in Chapter 4 and 6). However, over the three-year study period, the size structure and catch rates of lobsters in 2017 generally reflected that of the pre-construction survey in 2013 (Chapter 3). This indicates that post-construction the presence of wind farms had little observable effect on the lobster population in the area. Therefore, allowing fishing in OWF sites, can aid in the co-location of the two industries without detrimental effects to either industry. This thesis only reports on the interaction between an OWF development and static gear fisheries, the interaction with mobile gear fisheries are likely to have different effects observed.

Technological advances have allowed for offshore wind developments to be constructed further offshore. Energy generated from wind farms further offshore may be reduced during the transport to shore. To offset this loss, developers may have to install boosting stations along the export cable. For example, the Hornsea Four Project is proposing up to three boosting stations along the export cable to offset potential energy loss (Ørsted, 2019b). The addition of boosting stations increases the overall footprint of the development and further creates areas that may exclude commercial fishers. These additional structures in the marine environment may enhance biodiversity by providing new hard substrata for colonising species and refuge areas for fish species (Lacroix and Pioch, 2011; Coates *et al.*, 2014, 2016; Stenberg *et al.*, 2015; Vandendriessche *et al.*, 2015). They may also, together with the increased presence of turbines, aid in the spread of non-native species and the ranges of indigenous species due to their potential to act as stepping-stones for marine species (Petersen and Maim, 2006; Coates *et al.*, 2014; van Hal *et al.*, 2017; Dannheim *et al.*, 2018).

To date, most of offshore wind developments consist of turbines being fixed via different mechanisms to the seabed. These include monopiles, concrete and steel foundations, tripod foundations and tetrapod suction caissons, however monopiles are the predominant foundation type (Byrne *et al.*, 2002; Rodmell and Johnson, 2002; Bhattacharya *et al.*, 2015; Dannheim *et al.*, 2019b (Figure 4.1)). This is due to the developments being constructed in shallow, near shore areas. However, in areas where the shelf edge is close to shore, floating turbines may be used (Global Wind Energy

Council, 2015). For example, of the estimated 907 GW of offshore wind energy available in the United States, only 10% is estimated to be in water depths to allow traditional turbine foundations (Musial and Butterfield, 2004). Floating turbines allow for wind farms to be constructed in deeper waters with potentially less disruption to the marine environment due to the absence of pile driving and sand scour associated with traditional turbine foundations. Due to the securing requirements of floating turbines, the spatial extent of a possible exclusion to commercial fisheries of an individual floating turbine could be far greater than that of an individual traditional turbine. The floating turbines are required to be anchored to the seabed and have moorings ascending to the turbine. The moorings may need to be of a length up to 8 times the depth of the water and can have a spatial footprint of 1.13 – 4.52 km^2 (Statoil, 2015). The moorings may overlap adjacent floating turbines allowing for a spatial footprint to a similar sized development deploying turbines with traditional foundations. However, the presence of moorings and associated anchors may prevent both mobile and static fishing gear types operating between the turbines as is currently desirable and practiced in many OWFs. The risk of gear becoming entangled with the moorings may deter many fishers from fishing between floating turbines from both a safety and economic perspective due to potential gear loss. The Hywind development in Scotland, the first to deploy floating turbines (n= 5), may restrict mobile and static fisheries operating within the turbine array for the lifetime of the development (Statoil, 2015). The vertical and lateral movements of the mooring may cause physical scouring of the strata that may outweigh that generated by sand scour associated with traditional turbines (Skaare *et al.*, 2015). Deconflicting these issues, amongst others, is key to the two sectors co-existing.

The Fisheries Liaison with Offshore Wind and Wet Renewables (FLOWW) (overseen by the UK Crown Estates) is a collection of stakeholders, comprising both commercial fisheries and offshore renewable developers and forms part of the formal stakeholder consultation process (The Crown Estate, 2019). FLOWW was initiated, amongst other reasons, in response to a lack of understanding of offshore wind developers of the effects of developing OWFs on the fishing industry. FLOWW acts as a mechanism that allows best practice guidelines to be reached between the two industries, ranging from construction protocols for developers and guidelines on compensation agreements between the two industries. FLOWW produces guidelines based on lessons learnt from previous developments and considers future issues that may arise.

Within the UK, the FLOWW mechanism allows for the fishing industry to engage with the developer and work towards co-existence/co-location of the two industries. Contributions to FLOWW are made from commercial fisheries such as the National Federation of Fishermen’s Organisation (NFFO), Scottish Fishermen’s Federation (SFF) and the

Shellfish Association of Great Britain (SAGB). Government agencies such as the MMO and the Marine and Coastguard Agency are also FLOWW members, offering guidance on marine planning and marine navigational safety. Guidelines generated by FLOWW can aid both sectors in developing best practices for deconflicting issues and also aid stock and marine spatial managers in guiding their decisions.

7.3.2 Perspective of fishers

The development of an energy source that was CO^2 neutral (post construction) in comparison to oil and gas installations (Punt *et al.*, 2009) led to an impetus to develop the offshore wind industry. The impacts of these developments on the local ecology and fishing communities was not widely understood (Lindeboom *et al.*, 2011). The initial resistance from many local fishing communities was unexpected by some developers, leading to a requirement for a re-evaluation in the way the developers had worked with local fisheries. Whilst there is a need and drive within the scientific community to understand the ecological effects of offshore wind developments (Gill, 2005; Hooper and Austen, 2014b; Lüdeke, 2018; Dannheim *et al.*, 2019a), unpublished comments with various fishing communities in the UK and USA highlighted that many fishing communities do not share the same concerns (Karl Price, pers. comm. Skipper, *F.V. Isobella M*). Displacement and spatial conflict are the primary concerns of many fisheries of different gear types. The closure of the WMR wind farm displaced many of the inshore fishermen from their traditional grounds, this was a development with a relatively small spatial extent of 35 km^2. As offshore wind expands the overall spatial footprint is also increasing due to an increased presence of individual turbines, for example the Hornsea Zone projects (~ 50 km off the Holderness Coast), if planning permission is granted for all projects, have a combined spatial footprint of 2,165 km^2, an area 37% larger than London (Courtney French, pers. comm. Senior Environment & Consents Specialist, Ørsted). In comparison, a single gas platform, inclusive of the safety exclusion area of 500 m which is in force for the operational lifetime of the platform, has a minimum spatial footprint of approximately 0.785 km^2. Offshore wind turbines in UK waters do not operate a safety exclusion zone of 500 m around individual turbines or the whole wind farm. There is only an advised safe distance of 50 m from individual turbines which can be applied for by the developer from the MMO from construction to decommissioning (Dong Energy Ltd., 2009). A temporary 500 m exclusion safety zone is only enforced during the construction and maintenance of individual turbines. However, their spatial footprint is focused in specific areas whereas oil and gas terminals can be positioned widely throughout a fishery.

The very nature of the construction of offshore developments excludes other marine users for safety reasons. This displacement, in regard to commercial fishers, causes them to relocate to other available areas. This can be observed when marine exclusions are implemented for various reasons such as an MPA (Hart and Johnson, 2002a; Wiber *et al.*, 2012). For static gear fisheries (such as fixed nets and potting) which depend on retaining territory (Acheson, 1997; Hart *et al.*, 2002; Acheson and Gardner, 2011), this can cause a loss of income due to the time needed to relocate gear and also generate spatial conflict within the fishery as different fishers are trying to occupy smaller areas. It may also cause greater interactions between gear types, displacing mobile gear into areas that static gear operates. Hart and Johnson (2005) observed that different fishing sectors can organise agreements between sectors for spatial access between fisheries, as observed with the South West Potting Agreement (Devon & Severn IFCA, 2018). However increased spatial pressure from offshore developments can reduce or negate informal agreements and norms due to the reduced area available to different fishing sectors. For example, an informal agreement between a static and mobile gear fishery to deconflict gear interactions may be dissolved or ignored due to loss of available ground to fish. Prior to construction, there are often requirements for fishers to be excluded from areas due to surveys being conducted (e.g. geophysical) to aid in the planning process for the wind farm (Jamie Robertson, pers. comm. CEO, HFIG). As many of such surveys are required, this can lead to frustration from commercial fishers before construction is even approved. A common practice, for many offshore developers in the UK is to compensate the individual fishers for their displacement during construction periods and surveys, this is predominantly to maintain an area clear of fishing effort. However, many of these policies, whilst following guidelines from both sectors, do not account for the costs associated with being displaced (fuel costs of moving gear and increased competition in areas fishers are displaced too), just for maintaining clearance from an area and fishers can feel they have to cooperate irrespective of their own agenda (Suddaby, 2002). There is also the risk of inadvertently increasing effort on stocks as compensation is often reinvested in the fishers' business, for example, by purchasing more pots. Unpublished comments with UK and USA fishers (Karl Price, pers. comm. Skipper, *F.V. Isobella M*) has highlighted that many fishermen feel they are "second class citizens" with regards to marine spatial planning and they have to agree to cooperate as the development is happening anyway, irrespective of their concerns (Caveen *et al.*, 2014). However, engagement of individual fishing businesses in stakeholder consultation is traditionally quite low (as observed in the MCZ consultation process, DEFRA, 2019a) and the understanding of stakeholder organisations such as FLOWW are not widely understood in the fishing community.

There has historically been a reluctance within fishing communities to engage with scientific research either in data collection or acceptance of the results of scientific study, this can lead to a distrust of scientific research and subsequent management decisions (Mackinson and Nottestad, 1998; Mackinson *et al.*, 2006). Therefore, it may be easier for fishers to focus on the more tangible/ practical problems of displacement and navigation rather than the ecological effects of offshore wind development. Organisations such as Fishing into the Future (https://www.fishingintothefuture.co.uk), aim to increase the understanding of the wider fishing community with regards to science-based policy and management. There is a need for further engagement and understanding from the fishing industry of scientific process and management decisions. Initiatives using both scientific processes and a fisher's knowledge have been demonstrated to enhance the understanding of several fisheries (Zukowski *et al.*, 2011; Hind, 2014; Enever *et al.*, 2017), demonstrating a cooperation between scientific research and commercial fisheries.

7.4 Management suggestions

This thesis has highlighted that the construction and subsequent presence of an offshore wind farm was not detrimental to the local population of an inshore lobster fishery with regards to their size structure and catch rates and the invertebrate and fish bycatch in the area. Closure of the site to fishing exploitation during the construction period allowed larger lobsters to avoid capture, thus increasing the egg yield in the site and potentially benefitting the wider fishery via increasing larval production. There was a detrimental effect that the greater abundance of larger lobsters had on the other commercial species in the area, resulting in reduced catch rates of edible crab during the period. Whilst fishing restarted immediately post reopening of the site, reducing catch rates and size of lobsters, levels of effort and catch rates of lobsters reflected those of the control areas within the remaining six-week period of the survey.

Areas closed to fishing exploitation have been demonstrated to improve stock status of target species and overall biodiversity (Hoskin *et al.*, 2011; Lindeboom *et al.*, 2011; Moland *et al.*, 2013b). The use of OWFs, with their easily delineated boundaries, to act as a quasi-marine park may be a suitable management tool for marine managers. As there are already spatial restrictions to many forms of fishing types within wind farms, it may be beneficial to use these areas as some form of protected area. This may offset the need for additional MPAs in the area as the ecological benefits of wind farms and their introduction of new substratum types may enhance the areas biodiversity more than a traditional form of MPA (Coates *et al.* 2014; Coates *et al.* 2016; Krone *et al.* 2013a;

Krone *et al.* 2013b). This suggested multi-use of OWFs may enhance marine spatial planning processes whilst reducing the displacement of fishers in areas that occupy the same area as MPAs and offshore wind developments. However, some justifications for designation of an MPA/MCZ, for example, to preserve habitat conditions, may not overlap with the development of an OWF. In addition, post-construction, many fishers may wish to target OWFs and their potential stock enhancement or biodiversity benefits, this may cancel out any fishery enhanced benefits of exclusion during the construction phase.

The evidence supporting benefits of a closed area with regards to crustacean stocks such as lobsters is conflicting. A closed area can see an increase in abundance and biomass of lobsters as observed in this study and those of Hoskin *et al.* (2011) and Moland *et al.* (2013), resulting in a greater biomass of lobsters. However, the duration of the closure period may affect the biodiversity, abundance and density of lobsters in the closed area, due to potential increased competition for resources and increased chances of disease transmission (Wootton *et al.*, 2012; Davies *et al.*, 2015). If the development is in areas that support spawning stock or recruits to a fishery, this may benefit the wider fishery due to larval/recruit overspill into the wider area (Jessopp and McAllen, 2007). As observed in this study, protecting breeding stock from fishing exploitation, increases the egg yield from a closed area. However, alternative management measures (national ban on landing ovigerous lobsters (DEFRA, 2017) that have subsequently been introduced since this study may have a larger ecological impact on the egg yield within a fishery than that of a closed area. An OWF acting as an MPA/NTZ needs to consider these factors. Wootton *et al.*, (2012) and Davies *et al.*, (2015) both reported an increase in disease and occurrences of injury in lobsters caught in the Lundy island NTZ due to higher densities of larger lobsters when compared to a comparison site where fishing was taking place, If the closure is permanent, the potential benefit of closed areas to crustacean stocks such as increased larval production and recruitment may actually be outweighed by the potential detrimental effects to crustacean populations such as a reduction in overall biodiversity and increased incidences of disease.

An area could be closed for an optimum period that accounts for the brood cycle of the target species, allowing for increased egg yield from an area. The area could then be subsequently reopened to fishing exploitation to allow fishers to recover some of their economic losses due to the closure. Modelling of the optimum period of a closed area and the timing of subsequent re-opening of the site could be the focus of further study into the closed area effects. Closing areas and reopening the site for stock management has been common practice in other fisheries, rotational harvest has been demonstrated to be a suitable or proposed management tool (Hart, 2009; Cohen and Foale, 2013a;

Cohen *et al.*, 2013a; Lambert *et al.*, 2015). This study had highlighted that reopening of an area to fishing exploitation that was previously acting as an NTZ, was not detrimental to the local stocks within a relatively short time period of the opening. However, the reopening of the site should consider the “race to fish”, an example of the tragedy of the commons described by Hardin, (1968). The fishers that are capable of maximising on the economic return of opening the site are not necessarily the same fishers that were affected the most by the closure. Reopening of the site may need to be staggered to allow those most affected by the closure the opportunity to recuperate their losses first. Evidence to support this may be difficult to collate and may need vessel monitoring systems for smaller vessels than are currently legislated for. If a fishery overlaps with several offshore wind developments, theoretically a rotational closure system could be implemented to aid with stock management. This could also be implemented with regards to the aim for a coherent network of inter-connected MPA/MCZ/NTZs, described by the Marine Strategy Framework Directive (MFSD) (European Parliament and Council of the European Union, 2008) and the Maritime Spatial Planning Directive (MSPD) (The European Parliament and Council of the European Union, 2014).

A wind farm site could be used as a closed area, whether during the construction phase or as a site for a protected area. If the duration of the closure was for an optimum period, this may allow suitable protection of breeding stocks, potentially offsetting possible detrimental effects of closed areas. Reopening a previously closed area can allow for fishers to recuperate some of their losses due to exclusion from areas, which may encourage both compliance and engagement from the fishing industry. The construction period may be of a sufficient timescale to see some of the benefits of a closed area as observed in this study and further closure may not be desired. However, if this is the only closure period, any benefits observed will only be for a short period at the start of the potential 25-year lifecycle of the development and may not enhance the long-term sustainability of the species protected during the closure. The selection of sites and how they were used would need to be assessed on a case by case basis to maximise the benefits on the target species to be protected (Hilborn *et al.*, 2004).

Assessing multi-use of OWF sites could allow for an ecosystem approach with regards to marine spatial planning, meeting criteria for marine protection of crustacean stocks whilst also considering needs of commercial fishers considering the recommendations for requirements identified in Table 7.1.

Table 7.1: Suggestion for management and scientific requirements for the presence of offshore wind farms and their potential use as a management tool.

	Management implications		Scientist implication	
	Stock management	**Data collection initiatives**	**Stock management**	**Data collection initiatives**
Development of offshore wind farms	Spatial restrictions to marine safety and navigation.	Changes to shipping may affect stock migrations.	Mapping of new shipping and transit routes needed.	Dissemination of new shipping routes
	Spatial conflict with commercial fisheries and other marine users	Displacement of fishing effort may increase effort on stocks in other areas	Vessel monitoring systems fitted for all affected vessels (irrespective of size)	Spatial mapping of fishing effort changes in relation to the presence of wind turbines
	Creation of new habitat.	Potential to increase biodiversity in the area. Potential for stepping-stone effect for migration/introduction of non-resident species	Monitoring projects to monitor resident and new species of all associated flora and fauna	Time series analysis investigating the ecological changes associated with offshore wind farms following a paired series or BACI-PS approach
Wind farms as a permanently closed area to fishing exploitation	Spatial restrictions to marine safety and navigation.	Changes to shipping may affect stock migrations.	Mapping of new shipping and transit routes needed.	Dissemination of new shipping routes
	Site used as part of a network of marine protected areas.	Offer relief for target and non-target species from fishing exploitation. Potential increase in biodiversity and abundance of target and non-target species.	Long term monitoring project of the effects of the closed area and cumulative effects when used as part of a marine protected area network.	Time series analysis investigating the cumulative ecological effects of using an offshore wind farm as part of a network of marine protected areas following a BACI or BACI-PS approach
	Fishing effort displaced.	Reduction in fishing effort has a positive effect on marine fauna. Displacement of fishing effort may increase effort on stocks in other areas. Permanent economic loss to affected fishing industry.	Long term monitoring project investigating the effects of cessation of fishing effort has on marine fauna in the site and compare to surrounding areas where fishing effort is still present. Investigation into the socio-economic costs to the fishing industry associated with being permanently excluded from an area and wider stock implications	Spatial mapping of fishing effort changes in relation to the presence of wind turbines. Time series analysis investigating the cumulative ecological effects of using an offshore wind farm as part of a network of marine protected areas Economic analysis of the benefits of a permanently closed area as part of a network of protected areas.

Table 7.1. Cont.	Management implications		Scientist implication	
	Stock management	**Data collection initiatives**	**Stock management**	**Data collection initiatives**
Wind farms as a temporary closed area during construction	Spatial restrictions to marine safety and navigation.	Changes to shipping may affect stock migrations.	Mapping of new shipping and transit routes needed.	Dissemination of new shipping routes
	Site cannot be used as part of a network of marine protected areas due to the construction period being temporary	Offer temporary relief for target and non-target species from fishing exploitation. Potential increase in biodiversity and abundance of target and non-target species	Short term monitoring project of the effects of the closed area and compared to pre-construction surveys and control sites.	Temporal and spatial analysis investigating the ecological effects of cessation of fishing effort due to wind farm construction – comparison to pre-construction data and control sites
	Fishing effort displaced.	Temporary reduction in fishing effort can have a short-term positive effect on marine fauna. Temporary displacement of fishing effort may increase effort on stocks in other areas for the closure period. Temporary economic loss to affected fishing industry, potential to recuperate losses on opening of the site	Short term monitoring project investigating the short-term effects of cessation of fishing due to wind farm construction Investigation into the socio-economic costs to the fishing industry associated with short-term exclusion from an area and the wider stock implications of short-term absence of fishing mortality	Spatial mapping of fishing effort changes due to temporary exclusion. BACI or BACI-PS analysis investigating short-term exclusion effects Economic analysis of the benefits and costs of a temporary closed area to the wider system.

7.5 Limitations and further research

The results presented in this thesis and the conclusions drawn from such should consider the limitations of the study. Whilst a large number of individual animals were recorded, they were only sampled from two locations and over the summer period. Further sample stations both in the wind farm and surrounding area and over a greater period within the year would have given a more accurate portrayal of the effects of the construction of the WMR OWF and the closure of the site during the construction period. The study focussed primarily on the lobster population in the area via the use of mackerel as bait, other bait uses such as salmon heads (used to attract edible crabs) may have given a greater understanding of the bycatch in the area. Survey days were limited by the constraints of working on a vessel in the North Sea and the challenges of at sea data collection, subjected to changeable weather conditions and the stresses of mechanical failure of the vessel. The data gathered were also collated in a way to reflect the requirements of a monitoring project commissioned for the licensing agreement of the developer. Whilst this was planned by a steering committee consisting of the developer, independent scientists and the Holderness Fishing Industry Group, considering the resources available, a greater number of sites over all seasons using different bait types would have given a more accurate portrayal of the short-term effects of the development.

The thesis reports on the ecological benefits to a lobster population when a wind farm was closed to fishing exploitation during the construction phase. However, the socio-economic implications were not studied. Further investigation into the socio-economic implications of closing an area to a fishery that has specific site fidelity would be needed to investigate if the economic costs to a fishery are benefited by a closed area. Additionally, the socio-economic implications of reopening a site (to test the rotational harvest concept) would need to be investigated with regards to the tragedy of the common's paradigm.

Accurate estimations for effective trapping area and fecundity for the lobster population associated with the Westermost Rough OWF would give a greater understanding of the implications of the closed area with regards to egg yield. Catch coefficients of potting fisheries, accounting for factors affecting species retention in pots would also further enhance population and density estimates derived from catch statistics.

The results reported focused on the first three years post build of the wind farm. This only allows short-term effects of the development to be observed. A longer-term study, if commissioned could assess if there are any longer-term effects of offshore wind developments on commercially important crustacean species. The study demonstrates the importance of designing a monitoring programme that uses the uses empirical data to assess treatment effects when compared to a control site over a period of time following a paired series approach (suggested by Franco *et al.*, (2015)) and also the fishing industry and wind farm developer collaborating in the research.

7.6 Conclusions

To date, and the author's knowledge, this is the first empirical study investigating the effects of offshore wind developments on commercially important crustacean species. The principle aim of the thesis was to understand the short-term effects of construction of an offshore wind farm development on lobsters and associated bycatch was achieved. Enabling acceptance of the hypotheses that the size structure and catch rates of lobsters and commercial and non-commercial bycatch will be affected by the construction of the Westermost Rough offshore wind farm. Whilst initially there were concerns from fishers, that the development of offshore wind farms may be detrimental to crustacean stocks, this was not observed within the scope of this study. The secondary aim of the thesis, to investigate the effect of a temporary closure of the wind farm site during construction had on the ecology of the lobster and commercial and non-commercial bycatch was achieved, and highlighted the dominant changes observed within the study. There were additional benefits observed due to the closure of the site during the construction period, a positive response to the absence of fishing mortality on size structure, catch rates and total egg yield of lobsters was observed but was not observed for commercial bycatch species. This led to a partial acceptance of the hypotheses that the closure effect would have a positive effect on the lobster and commercial bycatch at the wind farm site. The importance of understanding these effects are essential to fishing communities and offshore wind developers as the expansion of offshore wind increases. Whilst the study area of the Holderness is the most important lobster fishery in Europe, it is still relatively small with regards to American lobster fisheries. These fisheries are currently facing the concerns that the Holderness fishers faced several years ago, with several large offshore wind developments currently being developed or proposed. To date there are only 5 turbines operational in US waters (Block Island wind farm). The results reported and conclusions drawn within this thesis may aid in the planning and understanding of the ecological effects of these future developments.

The study has also highlighted the importance of collaboration between the fishing industry and offshore wind developers. Using a fishing industry owned research vessel and employees to conduct the research allowed the fishery to engage with the science being conducted. The study was accepted by both the developer and the fishing industry and was accepted as the monitoring project report by the MMO, CEFAS and Natural England. There was also a sense of ownership of the project and the results gathered that offset some of their concerns. It also allowed the developer to formulate a relationship with the fishery, further increasing dialogue between the two industries to better understand the concerns of both parties. The results of this thesis and the way in which the study was approached may help other fisheries in their interactions with offshore wind developers.

Chapter 8: References

Abernethy, K. E., Trebilcock, P., Kebede, B., Allison, E. H., and Dulvy, N. K. 2010. Fuelling the decline in UK fishing communities? ICES Journal of Marine Science, 67: 1076–1085.

Acheson, J., and Gardner, R. 2011. The evolution of the Maine lobster V-notch practice: Cooperation in a prisoner's dilemma game. Ecology and Society, 16. (1): 41.

Acheson, J. M. 1997. The politics of managing the maine lobster industry: 1860 to the present. Human Ecology, 25: 3–27.

Acheson, J. M., and Taylor, L. 2001. The anatomy of the maine lobster co-management law. Society and Natural Resources, 14: 425–441.

Addison, J. T. 1995. Influence of behavioural interactions on lobster distribution and abundance as inferred from pot-caught samples. ICES mar. Sci. Symp, 199: 294–300.

Aiken, D. E., and Waddy, S. L. 2009. Controlling Growth And Reproduction In The American Lobster. *In* Proceedings of the annual meeting - World Mariculture Society, pp. 415–430.

Anderson, C. N. K., Hsieh, C. H., Sandin, S. A., Hewitt, R., Hollowed, A., Beddington, J., May, R. M., Sugihara, G. 2008. Why fishing magnifies fluctuations in fish abundance. Nature, 452: 835–839.

Arnason, R. 2007. The economics of rising fuel costs and European fisheries. EuroChoices, 6: 22–29.

Atema, J., Jacobsen, S., Karnofsky, E., Oleszko-Szuts, S., and Stein, L. 1979. Pair formation in the lobster, *Homarus americanus*: Behavioral development pheromones and mating. Marine and Freshwater Behaviour and Physiology, 6: 277–296.

Bailey, H., Senior, B., Simmons, D., Rusin, J., Picken, G., and Thompson, P. M. 2010. Assessing underwater noise levels during pile-driving at an offshore windfarm and its potential effects on marine mammals. Marine Pollution Bulletin, 60: 888–897.

Ball, B., Linnane, A., Munday, B., Browne, R., and Mercer, J. P. 2001. The effect of cover on in situ predation in early benthic phase European lobster, *Homarus gammarus*. Journal of Marine Biological Association, 81: 639–642.

Bannister, R. C. A., and Addison, J. T. 1998. Enhancing lobster stocks: A review of recent European methods results and future prospects. Bulletin of Marine Science, 62: 369–387.

Bannister, R.C.A, and Addison, J. T. 1998. Enhancing lobster stocks : a review of recent european methods , results , and future prospects. Bullitin of Marine Science, 62: 369–387.

Bates, D., Mächler, M., Bolker, B. M., and Walker, S. C. 2015. Fitting linear mixed-effects models using lme4. Journal of Statistical Software, 67: 1–48.

Belkin, I. M. 2009. Rapid warming of Large Marine Ecosystems. Progress in Oceanography, 81: 207–213.

Bell, M. C., Addison, J. T., and Bannister, R. C. A. 2001. Estimating trapping areas from trap-catch data for lobsters and crabs. Marine and Freshwater Research, 52: 1233–1242.

Bennett, D. B. 1974. The effects of pot immersion time on catches of crabs, *Cancer pagurus* L. and lobsters, *Homarus gammarus* (L.). Journal du Conseil Permanent International pour l'Exploration de la Mer, 35: 332–336.

Bennett, D. B. 1995. Factors in the life history of the edible crab (*Cancer pagurus* L.) that influence modelling and management. ICES Marine Science Symposium, 199: 89–98.

Bergman, M. J. N., Ubels, S. M., Duineveld, G. C. A., and Meesters, E. W. G. 2014. Effects of a 5-year trawling ban on the local benthic community in a wind farm in the Dutch coastal zone. ICES Journal of Marine Science, 71: 1088–1099.

Bertelsen, R. D., and Hornbeck, J. 2009. Using acoustic tagging to determine adult spiny lobster (*Panulirus argus*) movement patterns in the Western Sambo ecological reserve (Florida, United States). New Zealand Journal of Marine and Freshwater Research, 43: 35–46.

Bhattacharya, S., Cox, J., Lombardi, D., and Muir Wood, D. 2015. Dynamics of offshore wind turbines supported on two foundations. Geotechnical Engineering, 166: 159–169.

Blyth, R. E., Kaiser, M. J., Edwards-Jones, G., and Hart, P. J. B. 2002. Voluntary management in an inshore fishery has conservation benefits. Environmental Conservation, 29: 493–508.

Boudouresque, C., Astruch, P., Changeux, T., Ruitton, S., and Thibaut, T. 2019. Marine protected areas : multi-use management (MUM) vs . No-take zones (NTZ) and the efficiency of locally managed artisanal fishery. HAL-archives-ouvertes.fr.

Boudreau, S. A., and Worm, B. 2010. Top-down control of lobster in the Gulf of Maine : insights from local ecological knowledge and research surveys. Marine Ecology Progress Series, 403: 181–191.

Brandt, M. J., Diederichs, A., Betke, K., and Nehls, G. 2011. Responses of harbour porpoises to pile driving at the Horns Rev II offshore wind farm in the Danish North Sea. Marine Ecology Progress Series, 421: 205–216.

Branford, J. R. 1978. Incubation period for the lobster *Homarus gammarus* at various temperatures. Marine Biology, 47: 363–368.

Brčić, J., Herrmann, B., Mašanović, M., Baranović, M., Šifner, S. K., and Škeljo, F. 2018. Size selection of *Nephrops norvegicus* (L.) in commercial creel fishery in the Mediterranean Sea. Fisheries Research, 200: 25–32.

Byrne, B., Houlsby, G., Martin, C., and Fish, P. 2002. Suction Caisson Foundations for Offshore Wind Turbines. Wind Engineering, 26: 145–155.

Caddy, J. F. 1986. Modelling stock - recruitment processes in Crustacea: some practical and theoretical perspectives. Canadian Journal of Fisheries and Aquatatic Science, 43: 2330–2344.

Caddy, J. F., and Stamatopoulos, C. 1990. Mapping Growth and Mortality Rates of Organisms onto a Perforated Surface : The Relevance of 'Cover' to the Carrying Capacity of Natural and Artificial Habitats. Estuarine, Coastal and Shelf Science, 31: 87–106.

Causon, P. D., and Gill, A. B. 2018. Linking ecosystem services with epibenthic biodiversity change following installation of offshore wind farms. Environmental Science and Policy, 89: 340–347.

Caveen, A. J., Fitzsimmons, C., Pieraccini, M., Dunn, E., Sweeting, C. J., Johnson, M. L., Bloomfield, H., *et al.* 2014. Diverging strategies to planning an ecologically coherent network of MPAs in the North Sea: the roles of advocacy, evidence and pragmatism in the face of uncertainty. Advances in marine biology, 69: 325–370.

Cefas. 2017. Lobster (*Homarus gammarus*) Cefas Stock Status Report 2017: 18.

CEFAS. 2018. WaveNet Interactive map. http://wavenet.cefas.co.uk/Map (Accessed 17 July 2018).

Christie, N., Smyth, K., Barnes, R., and Elliott, M. 2014. Co-location of activities and designations: A means of solving or creating problems in marine spatial planning? Marine Policy, 43: 254–261.

Cisneros-Montemayor, A. M., Sanjurjo, E., Munro, G. R., Hernández-Trejo, V., and Rashid Sumaila, U. 2016. Strategies and rationale for fishery subsidy reform. Marine Policy, 69: 229–236.

Clarke, K. R., and Gorley, R. . 2006. PRIMER v6: user manual/tutorial. PRIMER-E, Plymouth, UK.

Close, C. H., and Hall, G. B. 2006. A GIS-based protocol for the collection and use of local knowledge in fisheries management planning. Journal of Environmental Management, 78: 341–352.

Coates, D. a., Deschutter, Y., Vincx, M., and Vanaverbeke, J. 2014. Enrichment and shifts in macrobenthic assemblages in an offshore wind farm area in the Belgian part of the North Sea. Marine Environmental Research, 95: 1–12.

Coates, D. A., Kapasakali, D. A., Vincx, M., and Vanaverbeke, J. 2016. Short-term effects of fishery exclusion in offshore wind farms on macrofaunal communities in the Belgian part of the North Sea. Fisheries Research, 179: 131–138.

Cohen, P. J., Cinner, J. E., and Foale, S. 2013a. Fishing dynamics associated with periodically harvested marine closures. Global Environmental Change, 23: 1702–1713.

Cohen, P. J., and Foale, S. J. 2013a. Sustaining small-scale fisheries with periodically harvested marine reserves. Marine Policy, 37: 278–287.

Cohen, P. J., and Alexander, T. J. 2013. Catch Rates, Composition and Fish Size from Reefs Managed with Periodically-Harvested Closures. PLoS ONE, 8.

Cohen, P. J., Cinner, J. E., and Foale, S. 2013b. Fishing dynamics associated with periodically harvested marine closures. Global Environmental Change, 23: 1702–1713.

Coleman, M. T., Porter, J. S., and Bell, M. C. 2019. Investigating fecundity and egg loss using a non-invasive method of brooding in European lobster (*Homarus gammarus*). ICES Journal of Marine Science: icesjms/fsz055.

Coolen, J. W. P., Lengkeek, W., Van Der Have, T., and Bittner, O. 2019. Upscaling positive effects of scour protection in offshore wind farms: Quick scan of the potential to upscale positive effects of scour protection on benthic macrofauna and associated fish species. 1–34 pp.

Costello, M. J., and Ballantine, B. 2015. Biodiversity conservation should focus on no-take Marine Reserves. Trends in Ecology & Evolution, 30: 507–509.

Crown Estates. 2019. Asset Map. https://www.thecrownestate.co.uk/en-gb/our-places/asset-map/#tab-2 (Accessed 2 October 2019).

Dannheim, J., Beermann, J., Lacroix, G., De Mesel, I., Kerckhof, F., Schön, I., Degraer, S., Birchenough, S., Garcia, C., Coolen, J.W.P., Lindeboom, H.J. 2018. Understanding the influence of man-made structures on the ecosystem functions of the North Sea (UNDINE): 47.

Dannheim, J., Bergström, L., Birchenough, S. N. R., Brzana, R., Boon, A. R., Coolen, J. W. P., Dauvin, J.-C., De Mesel, I., Derweduwen, J., Gill, A. B., Hutchison, Z. L., Jackson, A. C., Janas, U., Martin, G., Raoux, A., Reubens, J., Rostin, L., Vanaverbeke, J., Wilding, T. A., Wilhelmsson, D., Degraer, S. 2019a. Benthic effects of offshore renewables: identification of knowledge gaps and urgently needed research. ICES Journal of Marine Science.

Dannheim, J., Degraer, S., Elliot, M., and Smyth, K. 2019b. Seabed communities. *In* Wildlife and Wind Farms, Conflicts and Solutions., pp. 64–85. Pelagic Publishing.

Davies, C. E., Johnson, A. F., Wootton, E. C., Greenwood, S. J., Clark, K. F., Vogan, C. L., and Rowley, A. F. 2015. Effects of population density and body size on disease ecology of the European lobster in a temperate marine conservation zone. ICES Journal of Marine Science, 72: 128–138.

De Backer, A., Van Hoey, G., Coates, D., Vanaverbeke, J., and Hostens, K. 2014. Similar diversity-disturbance responses to different physical impacts: Three cases of small-scale biodiversity increase in the Belgian part of the North Sea. Marine Pollution Bulletin, 84: 251–262.

De Mesel, I., Kerckhof, F., Norro, A., Rumes, B., and Degraer, S. 2015. Succession and seasonal dynamics of the epifauna community on offshore wind farm foundations and their role as stepping stones for non-indigenous species. Hydrobiologia, 756: 37–50.

de Rozarieux, N. A. 2014. Use of discards in bait. National Federation of Fishemens Organisation. York.

De Troch, M., Reubens, J. T., Heirman, E., Degraer, S., and Vincx, M. 2013. Energy profiling of demersal fish: a case-study in wind farm artificial reefs. Marine environmental research, 92: 224–33.

Debuse, V. J., Addison, J. T., and Reynolds, J. D. 1999a. Effects of breeding site density on competition and sexual selection in the European lobster. Behavioral Ecology, 14: 396–402.

Debuse, V. J., Addison, J. T., and Reynolds, J. D. 1999b. The effects of sex ratio on sexual competition in the European lobster. Animal Behaviour, 58: 973–981.

Deep Water Wind. 2019. Block Island Wind Farm. http://dwwind.com/project/block-island-wind-farm/ (Accessed 21 May 2019).

DEFRA. 2017. Catching or landing of berried lobsters and crawfish in England. https://www.gov.uk/government/publications/catching-or-landing-of-berried-lobsters-and-crawfish-in-england/catching-or-landing-of-berried-lobsters-and-crawfish-in-england (Accessed 17 July 2018).

DEFRA. 2019a. Marine conservation zone designations in England. https://www.gov.uk/government/collections/marine-conservation-zone-designations-in-england#2013-mcz-designation-orders-and-factsheets (Accessed 26 March 2020).

DEFRA. 2019b. Consultation on the proposals for the third tranche of Marine Conservation Zones. London.

Devon & Severn IFCA. 2018. Current Byelaws and Permit Conditions. https://www.devonandsevernifca.gov.uk/ (Accessed 21 May 2019).

Dong Energy Ltd. 2009. Westermost Rough Offshore Wind Farm: Non-Technical Summary. 16–20 pp.

edf. 2019. Offshore wind:developing renewable marine energies. https://www.edf.fr/en/the-edf-group/industrial-provider/renewable-energies/wind-energy/offshore-wind-power (Accessed 2 October 2019).

edf Renewables. 2019. Key facts about Blyth. https://www.edf-re.uk/our-sites/blyth (Accessed 2 October 2019).

Ellis, C., Hodgson, D., Daniels, C., Collins, M., and Griffiths, A. 2017. Population genetic structure in European lobsters: implications for connectivity, diversity and hatchery stocking. Marine Ecology Progress Series, 563: 123–137.

Ellis, C. D., Knott, H., Daniels, C. L., Witt, M. J., and Hodgeson, D. J. 2015. Geographic and environmental drivers of fecundity in the European lobster (*Homarus gammarus*). ICES Journal of Marine Science, 72: 91–100.

Émond, K., Sainte-Marie, B., and Gendron, L. 2010. Relative growth, life-history phases, and sexual maturity of American lobster (*Homarus americanus*). Canadian Journal of Zoology, 88: 347–358.

Enever, R., Lewin, S., Reese, A., and Hooper, T. 2017. Mapping fishing effort: Combining fishermen's knowledge with satellite monitoring data in English waters. Fisheries Research, 189: 67–76.

English, P., Masson, T. I., Backstrom, J. T., Tibbles, B. J., Mackay, A. A., Smith, M. J., and Mitchell, T. 2017. Improving Efficiencies of National Environmental Policy Act Documentation for Offshore Wind Facilities Case Studies Report. 217 pp.

Eriksson, H., and Byrne, M. 2015. The sea cucumber fishery in Australia's Great Barrier Reef Marine Park follows global patterns of serial exploitation. Fish and Fisheries, 16: 329–341.

European Commission. 2016. Proposal for a Directive of the European Parliament and of the Council on the promotion of the use of energy from renewable sources.

European Parliament. 2019. The common fisheries policy: origins and development. http://www.europarl.europa.eu/factsheets/en/sheet/114/the-common-fisheries-policy-origins-and-development (Accessed 26 June 2019).

European Parliament and Council of the European Union. 2008. Directive 2008/56/EC of the European Parliament and of the Council of 17 June 2008 establishing a framework for community action in the field of marine environmental policy (Marine Strategy Framework Directive).

Figueiredo, J., Penha-Lopes, G., Anto, J., Narciso, L., and Lin, J. 2008. Fecundity, brood loss and egg development through embryogenesis of *Armases cinereum* (Decapoda: Grapsidae). Marine Biology, 154: 287–294.

Fogarty, M. J., and Gendron, L. 2004. Biological reference points for American lobster (*Homarus americanus*) populations : limits to exploitation and the precautionary approach. Canadian Journal of Fisheries & Aquatic Sciences, 61: 1392–1403.

Food and Agricultural Organization of the United Nations. 2019. *Homarus americanus* and *Homarus gammarus*. http://www.fao.org/fishery/species/2648/en & http://www.fao.org/fishery/species/2648/en (Accessed 7 May 2019).

Fox, J., and Weisberg, S. 2011. An (R) Companion to Applied Regression. SAGEPublications, Inc, Thousand Oaks (CA).

Franco, A., Quintino, V., and Elliott, M. 2015. Benthic monitoring and sampling design and effort to detect spatial changes: A case study using data from offshore wind farm sites. Ecological Indicators, 57: 298–304.

Freeman, D. J., MacDiarmid, A. B., and Taylor, R. B. 2009. Habitat patches that cross marine reserve boundaries: Consequences for the lobster *Jasus edwardsii*. Marine Ecology Progress Series, 388: 159–167.

Gill, A. B. 2005. Offshore renewable energy: Ecological implications of generating electricity in the coastal zone. Journal of Applied Ecology, 42: 605–615.

Global Wind Energy Council. 2015. Global Wind Report 2015. 75 pp.

Gnanalingam, G., and Hepburn, C. 2015. Flexibility in temporary fisheries closure legislation is required to maximise success. Marine Policy, 61: 39–45. Elsevier.

Goldstein, J. S., and Watson III, W. H. 2019. Biochemical changes throughout early- and middle-stages of embryogenesis in lobsters (*Homarus americanus*) under different thermal regimes. PeerJ, 7: e6952.

Goni, R., Quetglas, A., and Renones, O. 2003. Size at maturity, fecundity and reproductive potential of a protected population of the spiny lobster *Palinurus elephas* (Fabricius, 1787) from the western Mediterranean. Marine Biology, 143: 583–592.

Goñi, R., Hilborn, R., Díaz, D., Mallol, S., and Adlerstein, S. 2010. Net contribution of spillover from a marine reserve to fishery catches. Marine Ecology Progress Series, 400: 233–244.

Grabowski, J. H., Clesceri, E. J., Yund, P. O., Weber, M., Poland, P., and Myrick, M. 2005. Are We Using Herring to Farm Lobsters? The Effect of Herring Bait on Lobster Growth, and the Fate of Discarded Bait on Bottom Habitat. Portland.

Grabowski, J. H., Clesceri, E. J., Baukus, A. J., Gaudette, J., Weber, M., and Yund, P. O. 2010. Use of herring bait to farm lobsters in the Gulf of Maine. PLoS ONE, 5.

Green, B. S., Gardner, C., Hochmuth, J. D., and Linnane, A. 2014. Environmental effects on fished lobsters and crabs. Reviews in Fish Biology and Fisheries, 24: 613–638.

Griffin, R., Buck, B., and Krause, G. 2015. Private incentives for the emergence of co-production of offshore wind energy and mussel aquaculture. Aquaculture, 436: 80–89.

Griffin, R. A., Robinson, G. J., West, A., Gloyne-Phillips, I. T., and Unsworth, R. K. F. 2016. Assessing fish and motile fauna around offshore windfarms using stereo baited video. PLoS ONE, 11: 1–14.

Gwinn, D. C., Allen, M. S., Johnston, F. D., Brown, P., Todd, C. R., and Arlinghaus, R. 2015. Rethinking length-based fisheries regulations: The value of protecting old and large fish with harvest slots. Fish and Fisheries, 16: 259–281.

Haig, J. A., Bakke, S., Bell, M. C., Bloor, I. S. M., Cohen, M., Coleman, M., Dignan, S., Kaiser, M.J., Pantin, J. R., Roach, M., Salomonsen, H. and Tully, O. 2016. Reproductive traits and factors affecting the size at maturity of *Cancer pagurus* across Northern Europe . ICES Journal of Marine Science: Journal du Conseil, 73: 2572–2585.

Hall, G. B., and Close, C. H. 2007. Local knowledge assessment for a small-scale fishery using geographic information systems. Fisheries Research, 83: 11–22.

Hanson, J. M., and Lanteigne, M. (n.d.). Evaluation of Atlantic cod predation on American lobster in the southern Gulf of St. Lawrence, with comments on other potential fish predators. Transactions of the American Fisheries Society, 129: 13–29. American Fisheries Society.

Hardin, G. 1968. The tragedy of ther commons. Science, 162: 1243–1248.

Harnish, L., and Willison, J. H. M. 2009. Efficiency of bait usage in the Nova Scotia lobster fishery: a first look. The Sustainability of Seafood Production and Consumption, 17: 345–347.

Hart, D. R. 2009. Improving utilization of the Atlantic sea scallop resource: an analysis of rotational management of fishing grounds: comment. Land Economics, 83: 378–382.

Hart, P., and Johnson, M. L. 2002. Who Owns the Sea. *In* Who Owns the Sea, pp. 1–105.

Hart, P. J. ., Blyth, R. B., Kaiser, M. J., and Edwards Jones, G. 2002. Sustainable exploitation with minimal conflict: is it possible? *In* Who Owns the Sea.

Hilborn, R., Stokes, K., Maguire, J. J., Smith, T., Botsford, L. W., Mangel, M., Orensanz, J., *et al.* 2004. When can marine reserves improve fisheries management? Ocean and Coastal Management, 47: 197–205.

Hind, E. J. 2014. A review of the past, the present, and the future of fishers' knowledge research: a challenge to established fisheries science. ICES Journal of Marine Science, 72: 341–358.

Hoare, D., Graham, N., and Schön, P. J. 2011. The Irish Sea data-enhancement project: Comparison of self-sampling and national data-collection programmes - Results and experiences. ICES Journal of Marine Science, 68: 1778–1784.

Holland, D., Gudmundsson, E., and Gates, J. 1999. Do fishing vessel buyback programs work: A survey of the evidence. Marine Policy, 23: 47–69.

Holland, D. S., Steiner, E., and Warlick, A. 2017. Can vessel buybacks pay off: An evaluation of an industry funded fishing vessel buyback. Marine Policy, 82: 8–15.

Holst, R., and Revill, A. 2009. A simple statistical method for catch comparison studies. Fisheries Research, 95: 254–259.

Hooper, T., and Austen, M. 2014. The co-location of offshore windfarms and decapod fisheries in the UK: Constraints and opportunities. Marine Policy, 43: 295.

Hooper, T., Ashley, M., and Austen, M. 2015. Perceptions of fishers and developers on the co-location of offshore wind farms and decapod fisheries in the UK. Marine Policy, 61: 16–22.

Hooper, T., Beaumont, N., and Hattam, C. 2017. The implications of energy systems for ecosystem services: A detailed case study of offshore wind. Renewable and Sustainable Energy Reviews, 70: 230–241.

Hoskin, M. G., Coleman, R. a., von Carlshausen, E., and Davis, C. M. 2011. Variable population responses by large decapod crustaceans to the establishment of a temperate marine no-take zone. Canadian Journal of Fisheries and Aquatic Sciences, 68: 185–200.

Howard, A. E. 1982. The distribution and behaviour of ovigerous edible crabs (*Cancer pagurus*), and consequent sampling bias. Journal du Conseil / Conseil Permanent International pour l'Exploration de la Mer, 40: 259–261.

Howarth, L. M., Dubois, P., Gratton, P., Judge, M., Christie, B., Waggitt, J. J., Hawkins, J. P., *et al.* 2017. Trade-offs in marine protection: Multispecies interactions within a community-led temperate marine reserve. ICES Journal of Marine Science, 74: 263–276.

Huserbråten, M. B. O., Moland, E., Knutsen, H., Olsen, E. M., André, C., and Stenseth, N. C. 2013. Conservation, Spillover and Gene Flow within a Network of Northern European Marine Protected Areas. PLoS ONE, 8.

International Council for the Exploration of the Sea. 2016. ICES WGCRAB REPORT 2013 Working Group on the Biology and Report of the Life History of Crabs (WGCRAB). 27–31 pp.

IUCN. 2008. Guidelines for Applying Protected Area Management Categories.

Jenkins, S. R., Beukers-Stewart, B. D., and Brand, A. R. 2001. Impact of scallop dredging on benthic megafauna: A comparison of damage levels in captured and non-captured organisms. Marine Ecology Progress Series, 215: 297–301.

Jessopp, M. J., and McAllen, R. J. 2007. Water retention and limited larval dispersal: Implications for short and long distance dispersers in marine reserves. Marine Ecology Progress Series, 333: 27–36.

JNCC. 2011. How many No Take Zones are either already operating or planned for UK waters? JNCC.

Johannes, R. E., Freeman, M. M. R., and Hamilton, R. J. 2008. Ignore fishers' knowledge and miss the boat. Fish and Fisheries, 1: 257–271.

Johnson, K. J., Goldstein, J. S., and Watson, W. H. 2011. Two Methods for Determining the Fertility Status of Early-Stage American Lobster, *Homarus americanus*, Eggs. Journal of Crustacean Biology, 31: 693–700.

Johnson, M. L., and Johnson, M. P. 2013. Advances in Marine Biology; The Ecology and Biology of *Nephrops norvegicus*.

Johnson, T. R., and Van Densen, W. L. T. 2007. Benefits and organization of cooperative research for fisheries management. ICES Journal of Marine Science, 64: 834–840.

Jones, G., Srinivasan, M., and Almany, G. 2007. Population Connectivity and Conservation of Marine Biodiversity. Oceanography, 20: 100–111.

Jørstad, K. E., Agnalt, A.-L., and Farestveit, E. 2011. The Introduced American Lobster, *Homarus americanus* in Scandinavian Waters. *In* In the Wrong Place - Alien Marine Crustaceans: Distribution, Biology and Impacts, pp. 625–638.

Jury, S. H., Howell, H., O'Grady, D. F., and Watson III, W. H. 2001. Lobster trap video: in situ video surveillance of the behaviour of *Homarus americanus* in and around traps. Marine and Freshwater Research, 52: 1125–1132.

Jury, S. H., and Watson, W. 2013. The relationship between American lobster catch, entry rate into traps and density. Marine Biology Research, 9: 59–68.

Kaiser, M. J. 2005. Are Marine Protected Areas a red herring or fisheries panacea? Canadian Journal of Fisheries and Aquatic Sciences, 62: 1194–1199.

Kamermans, P., Walles, B., Kraan, M., van Duren, L. A., Kleissen, F., van der Have, T. M., Smaal, A. C., andPoelman, M. 2018. Offshore wind farms as potential locations for flat oyster (*Ostrea edulis*) restoration in the Dutch North Sea. Sustainability (Switzerland), 10.

Karnofsky, E. B., and Price, H. J. 1989. Behavioura Response of the Lobster *Homarus americanus* to traps. Can. J. Fish. Aquat. Sci., 46: 1625-1632.

Kelley, K. H. 1992. A summary of Maine Lobster Laws and Regulations, 1820 - 1992. 3 pp.

Kjelland, M. E., Piercy, C. D., Lackey, T., and Swannack, T. M. 2015. An integrated modeling approach for elucidating the effects of different management strategies on Chesapeake Bay oyster metapopulation dynamics. Ecological Modelling, 308: 45–62.

Koopman, H. N., Westgate, A. J., and Siders, Z. A. 2015. Declining fecundity and factors affecting embryo quality in the American lobster (*Homarus americanus*) from the Bay of Fundy. Canadian Journal of Fisheries and Aquatic Sciences, 72: 352–363.

Köster, F. W., Stephenson, R. L., and Trippel, E. A. 2014. Gadoid fisheries: the ecology and management of rebuilding. ICES Journal of Marine Science, 71: 1311–1316.

Krone, R., Gutow, L., Joschko, T. J., and Schröder, A. 2013a. Epifauna dynamics at an offshore foundation – Implications of future wind power farming in the North Sea. Marine Environmental Research, 85: 1–12.

Krone, R., Gutow, L., Brey, T., Dannheim, J., and Schröder, a. 2013b. Mobile demersal megafauna at artificial structures in the German Bight – Likely effects of offshore wind farm development. Estuarine, Coastal and Shelf Science, 125: 1–9.

Krone, R., Dederer, G., Kanstinger, P., Krämer, P., Schneider, C., and Schmalenbach, I. 2017. Mobile demersal megafauna at common offshore wind turbine foundations in the German Bight (North Sea) two years after deployment - increased production rate of *Cancer pagurus*. Marine Environmental Research, 123: 53–61.

Lacroix, D., and Pioch, S. 2011. The multi-use in wind farm projects: more conflicts or a win-win opportunity? Aquatic Living Resources, 24: 129–135.

Lambert, G. I., Murray, L. G., Hiddink, J. G., Hinz, H., Salomonsen, H., and Kaiser, M. J. 2015. Impact of scallop dredging on benthic communities and habitat features in the Cardigan Bay Special Area of Conservation Part I – Impact on infaunal invertebrates. Fisheries and Conservation Report No. 59. 97 pp.

Le Bris, A., Mills, K. E., Wahle, R. A., Chen, Y., Alexander, M. A., Allyn, A. J., Schuetz, J. G., *et al.* 2018. Climate vulnerability and resilience in the most valuable North American fishery. Proceedings of the National Academy of Sciences, 115: 1831–1836.

Leal, M. C., Pochelon, P. N., Lopes da Silva, T., Reis, A., Rosa, R., and Calado, R. 2013. Variable within-brood maternal provisioning in newly extruded embryos of *Homarus gammarus* Variable within-brood maternal provisioning in newly extruded embryos of Homarus gammarus. Marine Biodiversity, 160: 763–772.

Lees, K., Pitois, S., Scott, C., Frid, C., and Mackinson, S. 2006. Characterizing regime shifts in the marine environment. Fish and Fisheries, 7: 104–127.

Leleu, K., Alban, F., Pelletier, D., Charbonnel, E., Letourneur, Y., and Boudouresque, C. F. 2012. Fishers' perceptions as indicators of the performance of Marine Protected Areas (MPAs). Marine Policy, 36: 414–422.

Leocádio, A. M., Whitmarsh, D., and Castro, M. 2012. Comparing trawl and creel fishing for Norway Lobster (*Nephrops norvegicus*): Biological and economic considerations. PLoS ONE, 7.

Lindeboom, H., Degraer, S., Dannheim, J., Gill, A. B., and Wilhelmsson, D. 2015. Offshore wind park monitoring programmes, lessons learned and recommendations for the future. Hydrobiologia, 756: 169–180.

Lindeboom, H. J., Kouwenhoven, H. J., Bergman, M. J. N., Bouma, S., Brasseur, S., Daan, R., Fijn, R. C., *et al.* 2011. Short-term ecological effects of an offshore wind farm in the Dutch coastal zone; a compilation. Environmental Research Letters, 6: 035101.

Link, J. S., Bogstad, B., Sparholt, H., and Lilly, G. R. 2009. Trophic role of Atlantic cod in the ecosystem. Fish and Fisheries, 10: 58–87.

Linnane, a. J., Penny, S. S., and Ward, T. M. 2008. Contrasting fecundity, size at maturity and reproductive potential of southern rock lobster *Jasus edwardsii* in two South Australian fishing regions. Journal of the Marine Biological Association of the UK, 88.

Lizárraga-Cubedo, H. a., Tuck, I., Bailey, N., Pierce, G. J., and Kinnear, J. a. M. 2003. Comparisons of size at maturity and fecundity of two Scottish populations of the European lobster, *Homarus gammarus*. Fisheries Research, 65: 137–152.

Long, S. 2017. Short-term impacts and value of a periodic no take zone (NTZ) in a community-managed small-scale lobster fishery, Madagascar. PLoS ONE, 12: 1–23.

Lovewell, S. R. . 1991. Observations on the by-catch of lobsters (*Homarus gammarus* L.) by trawlers off the Yorkshire coast, England. ICES Conference Meetings, K19.

Lovewell, S. R. J., Howard, A. E., and Bennett, D. B. 1988. The effectiveness of parlour pots for catching lobsters (*Homarus gammarus (L.)*) and crabs (*Cancer pagurus L.*). J. Cons. int. Explor. Mer, 44: 247–252.

Lovewell, S. R. J., and Addison, J. T. 1991. Size composition and pot selectivity in the lobster (*Homarus gammarus* (L.)) and crab (*Cancer pagurus* L.) fisheries on the east coast of England. ICES Journal of Marine Science, 482859: 79–90.

Lüdeke, J. 2018. Exploitation of Offshore Wind Energy. Handbook on Marine Environment Protection: 165–188.

Mackinson, S., and Nottestad, L. 1998. Combining local and scientific knowledge. Reviews in Fish Biology and Fisheries, 8: 481–490.

Mackinson, S. 2001. Integrating local and scientific knowledge: an example in fisheries science. Environmental management, 27: 533–545.

Mackinson, S., Curtis, H., Brown, R., Mctaggart, K., Taylor, N., Neville, S., and Rogers, S. 2006. A report on the perceptions of the fishing industry into the potential socio-economic impacts of offshore wind energy developments on their work patterns and income. Technical report 133. CEFAS, Lowerstoft.

Madsen, P. T., Wahlberg, M., Tougaard, J., and Tyack, P. 2006. Wind turbine underwater noise and marine mammals: implications of current knowledge and data needs. Mar. Ecol. Progr. Ser., 309: 279–295.

Marine Management Organisation. 2011. UK Sea Fisheries Statistics 2011.

Marine Management Organisation. 2017b. £28,220 penalty for release of non-native species. https://www.gov.uk/government/news/28220-penalty-for-release-of-non-native-species (Accessed 7 May 2019).

Marine Management Organisation. 2018. Provisional Results: Monthly UK Sea Fisheries Statistics - Reproted Landings November 2018. 21 pp.

Marine Management Organisation. 2019a. Monthly Sea Fisheries Statistics December 2019. https://www.gov.uk/government/statistics/monthly-sea-fisheries-statistics-december-2019 (Accessed 11 March 2020).

Marine Management Organisation. 2019b. Landings obligation general requirements 2019. https://www.gov.uk/government/publications/landing-obligation-2019-rules-and-regulations/landing-obligation-general-requirements-2019--2#fisheries-that-are-not-affected-by-the-landing-obligation (Accessed 26 June 2019).

Markus, T. 2010. Towards sustainable fisheries subsidies: Entering a new round of reform under the Common Fisheries Policy. Marine Policy, 34: 1117–1124.

Melville-Smith, R., and De Lestang, S. 2006. Spatial and temporal variation in the size at maturity of the western rock lobster *Panulirus cygnus* George. Marine Biology, 150: 183–195.

Mente, E., Houlihan, D. F., and Smith, K. 2001. Growth, feeding frequency, protein turnover, and amino acid metabolism in European lobster *Homarus gammarus*_(L) . J.Exp.Zool. , 289: 419–432.

Merder, J., Browne, P., Freund, J. A., Fullbrook, L., Graham, C., Johnson, M. P., Wieczorek, A.,and Power, A. M. 2019. Density-dependent growth in ‘catch-and-wait’ fisheries has implications for fisheries management and Marine Protected Areas. Ambio. 49: 1. 107-117.

Merideth, D. 1999. The strategic importance of the fishing sector to rural communities and Ireland: a case study of the Rossaveal Region, Co. Galway. Marine Institute.

Mills, K. E., Pershing, A. J., Brown, C. J., Chen, Y., Chiang, F. S., Holland, D. S., Lehuta, S., *et al.* 2013. Fisheries management in a changing climate. Oceanography, 26: 191–195.

Moland, E., Olsen, E. M., and Stenseth, N. C. 2010. Maternal influences on offspring size variation and viability in wild European lobster *Homarus gammarus*. Marine Ecology Progress Series, 400: 165–173.

Moland, E., Olsen, E. M., Knutsen, H., Knutsen, J. A., Enersen, S. E., André, C., and Stenseth, N. C. 2011a. Activity patterns of wild European lobster *Homarus gammarus* in coastal marine reserves: Implications for future reserve design. Marine Ecology Progress Series, 429: 197–207.

Moland, E., Olsen, E. M., Andvord, K., Knutsen, J. A., and Stenseth, N. C. 2011b. Home range of European lobster (*Homarus gammarus*) in a marine reserve : implications for future reserve design. Canadian Journal of Fisheries and Aquatic Sciences, 1210: 1197–1210.

Moland, E., Ulmestrand, M., Olsen, E. M., and Stenseth, N. C. 2013a. Long-term decrease in sex-specific natural mortality of European lobster within a marine protected area. Marine Ecology Progress Series, 491: 153–164.

Moland, E., Olsen, E. M., Knutsen, H., Garrigou, P., Espeland, S. H., Kleiven, A. R., Andre, C., *et al.* 2013b. Lobster and cod benefit from small-scale northern marine protected areas: inference from an empirical before-after control-impact study. Proceedings of the Royal Society : Biological Sciences, 280: 20122679–20122679.

Moller, H., Berkes, F., Lyver, P. O., and Kislalioglu, M. 2016. Combining Science and Traditional Ecological Knowledge: Monitoring Populations for Co-Management. Ecology and Society, 9.

Mori, M., Modena, M., and Biagi, F. 2001. Fecundity and egg volume in Norway lobster (*Nephrops norvegicus*) from different depths in the northern Tyrrhenian Sea, 32: 1641-1650.

Murawski, S. A., Brown, R., Lai, H. L., Rago, P. J., and Hendrickson, L. 2000. Large-Scale Closed Areas As a Fishery Tool in Temperate Marine Systems : the Georges Bank Experience. Bulletin of Marine Science, 66: 775–798.

Musial, W., and Butterfield, S. 2004. Future for Offshore Wind Energy in the United States. *In* Energy ocean.

NEIFCA. 2003. Trawling: Prohibition: Exceptions.

Nicoll, R., and Day, J. C. 2017. Correct application of the IUCN protected area management categories to the CCAMLR Convention Area. Marine Policy, 77: 9–12.

NOAA Fisheries. 2017. Annual Commercial Landings Statistics. https://www.st.nmfs.noaa.gov/commercial-fisheries/commercial-landings/annual-landings/index (Accessed 7 May 2019).

North Eastern Inshore Fisheries and Conservation Authority. 2015. Crustacea Conservation Byelaw.

O'Regan, S. M. 2015. Harvesters' perspectives on the management of British Columbia's giant red sea cucumber fishery. Marine Policy, 51: 103–110.

Obura, D. O., Wells, S., Church, J., and Horrill, C. 2002. Monitoring of fish and fish catches by local fishermen in Kenya and Tanzania. Marine and Freshwater Research, 53: 215–222.

Ogle, D. 2016a. Among Group Statistical Comparisons. *In* Introductory Fisheries Analyses with R, First, pp. 122–130. Ed. by D. Ogle. Taylor and Francis Group, Boca Raton.

Ogle, D. 2016b. Cumulative Frequencies. *In* Introductory Fisheries Analysis with R, First, pp. 111–112. Ed. by D. Ogle. Taylor and Francis Group, Boca Raton.

Öndes, F., Emmerson, J. A., Kaiser, M. J., Murray, L. G., and Kennington, K. 2017. The catch characteristics and population structure of the brown crab (*Cancer pagurus*) fishery in the Isle of Man, Irish Sea. Journal of the Marine Biological Association of the United Kingdom: 1–15.

Ørsted. 2019a. Our Wind Farms. https://orsted.co.uk/en/Generating-energy/Offshore-wind/Our-wind-farms (Accessed 21 May 2019).

Ørsted. 2019b. Hornsea Project Four. https://hornseaprojects.co.uk/Hornsea-Project-Four (Accessed 2 October 2019).

Parma, A. M., Hilborn, R., and Orensanz, J. M. L. 2006. The good, the bad and the ugly: learning from experience to achieve sustainable fisheries. Bulletin of Marine Science, 78: 411–427.

Pereira, G., and Josupeit, H. 2017. The world lobster market. Rome. 41 pp.

Petersen, J. K., and Maim, T. 2006. Offshore windmill farms: threats to or possibilities for the marine environment. Ambio, 35: 75–80.

Pettersen, A. R., Moland, E., Olsen, E. M., and Knutsen, J. A. 2009. Lobster Reserves in Coastal Skagerrak - An Integrated Analysis of the Implementation Process. Integrated Coastal Zone Management: 178–188.

Phillips, B. 2007. Lobsters, Biology, Management, Aquaculture and Fisheries.

Planque, B., and Fredou, T. 1999. Temperature and the recruitment of Atlantic cod (*Gadus morhua*).. Canadian Journal of Fisheries and Aquatatic Science, 56: 2069–2077.

Punt, M. J., Groeneveld, R. A., van Ierland, E. C., and Stel, J. H. 2009. Spatial planning of offshore wind farms: A windfall to marine environmental protection? Ecological Economics, 69: 93–103.

Ramsay, K., Kaiser, M. J., and Hughes, R. N. 1998. Responses of benthic scavengers to fishing disturbance by towed gears in different habitats. Journal of Experimental Marine Biology and Ecology, 224: 73–89.

Raoux, A., Tecchio, S., Pezy, J. P., Lassalle, G., Degraer, S., Wilhelmsson, D., Cachera, M., Ernande, B., Le Guen, C., Haraldsson, M., Grangere, K., Le Loc'h, F., Dauvin, J-C. and Niquil, N. 2017. Benthic and fish aggregation inside an offshore wind farm: Which effects on the trophic web functioning? Ecological Indicators, 72: 33–46.

Roach, M., Cohen, M., Forster, R., Revill, A. S., and Johnson, M. 2018. The effects of temporary exclusion of activity due to wind farm construction on a lobster (*Homarus gammarus*) fishery suggests a potential management approach. ICES Journal of Marine Science. 75:4. 1416-1426

Roach, M. 2019. Westermost Rough Shellfish Monitoring Report 2019: A Study Conducted for Ørsted. Bridlington. 22–23 pp.

Rodmell, D., and Johnson, M. 2002. The development of marine based wind energy generation and inshore fisheries in UK waters: Are they compatible? *In* Who owns the sea?, pp. 76–103.

Rötzer, M. A. I. N., and Haug, J. T. 2015. Larval development of the European lobster and how small heterochronic shifts lead to a more pronounced metamorphosis. International Journal of Zoology, 2015.

Russell, J., and Solutions, A. 2017. Analysis of nephrops industry in Scotland. Edinburgh. 25–47 pp.

Saila, S. B., Nixon, S. W., and Oviatt, C. A. 2002. Does Lobster Trap Bait Influence the Maine Inshore Trap Fishery? North American Journal of Fisheries Management, 22: 602–05.

Sala, E., and Giakoumi, S. 2018. No-take marine reserves are the most effective protected areas in the ocean. ICES Journal of Marine Science, 75: 1166–1168.

Sas, H., van der Have, T. M., Kamermans, P., and Lengkeek, W. 2018. Flat oyster pilot design in North Sea offshore wind farm: 1–35.

Schmalenbach, I., and Franke, H.-D. 2010. Potential impact of climate warming on the recruitment of an economically and ecologically important species, the European lobster (*Homarus gammarus*) at Helgoland, North Sea. Marine Biology, 157: 1127–1135.

Scottish Government. 2019. New Landings Controls for the Crab and Lobster Fisheries.

https://www2.gov.scot/Topics/marine/Sea-Fisheries/InshoreFisheries/crab-lobster-landing-controls (Accessed 1 April 2019).

Seafish. 2019. Pots and traps - lobster.

https://www.seafish.org/gear/gear/profile/pots-and-traps-lobster (Accessed 30 April 2019).

Sekhon, J. S. 2011. Multivariate and Propensity Score Matching Software with Automated Balance Optimization: The Matching Package for R. Journal of Statistical Software, 42: 1–52.

Sheehy, M. R. J., Bannister, R. C. A., Wickins, J. F., and Shelton, P. M. J. 1999. New perspectives on the growth and longevity of the European lobster (*Homarus gammarus*). Canadian Journal of Fisheries and Aquatic Sciences, 56: 1904–1915.

Sheehy, M. R. J., and Bannister, R. C. A. 2002. Year-class detection reveals climatic modulation of settlement strength in the European lobster, *Homarus gammarus*. Canadian Journal of Fisheries and Aquatic Sciences, 59: 1132–1143.

Sibert, V., Ouellet, P., and Brethes, J.-C. 2004. Changes in yolk total proteins and lipid components and embryonic growth rates during lobster (*Homarus americanus*) egg development under a simulated seasonal temperature cycle. Marine Biology, 144: 1075–1086.

Skaare, B., Nielsen, F. G., Hanson, T. D., Yttervik, R., Havmoller, O., and Rekdal, A. 2015. Analysis of measurements and simulations from the Hywind Demo floating wind turbine. Wind Energy, 18: 1105–1122.

Skog, M. 2009. Intersexual differences in European lobster (*Homarus gammarus*): recognition mechanisms and agonistic behaviours. Behaviour, 146: 1071–1091.

Small, D. P., Calosi, P., Boothroyd, D., Widdicombe, S., and Spicer, J. I. 2016. The sensitivity of the early benthic juvenile stage of the European lobster *Homarus gammarus* (L.) to elevated pCO2and temperature. Marine Biology, 163: 1–12.

Smith, I. P., Collins, K. J., and Jensen, a. C. 1998. Movement and activity patterns of the European lobster, *Homarus gammarus*, revealed by electromagnetic telemetry. Marine Biology, 132: 611–623.

Smith, I. P., Collins, K. J., and Jensen, A. C. 1999. Seasonal changes in the level and diel pattern of activity in the European lobster *Homarus gammarus*. Marine Ecology Progress Series, 186: 255–264.

Smith, I. P., Jensen, A. C., Collins, K. J., and Mattey, E. L. 2001. Movement of wild European lobsters *Homarus gammarus* in natural habitat. Marine Ecology Progress Series, 222: 177–186.

Smith, M. T., and Addison, J. T. 2003. Methods for stock assessment of crustacean fisheries. Fisheries Research, 65: 231–256.

Smolowitz, R. J. 1978. Lobster, *Homarus americanus*, Trap Design and Ghost Fishing. Marine Fisheries Review.

Smyth, K., Christie, N., Burdon, D., Atkins, J. P., Barnes, R., and Elliott, M. 2015. Renewables-to-reefs? – Decommissioning options for the offshore wind power industry. Marine Pollution Bulletin, 90: 247–258.

Sørdalen, T. K., Halvorsen, K. T., Harrison, H. B., Ellis, C. D., Vøllestad, L. A., Knutsen, H., Moland, E. 2018. Harvesting changes mating behaviour in European lobster. Evolutionary Applications, 11: 963–977. Wiley-Blackwell.

SSE Renewables & equinor. 2019. Dogger Bank Wind Farms. https://doggerbank.com/ (Accessed 2 October 2019).

State of Maine Department of Marine Resources. 2009. A guide to lobstering in Maine.

https://www.maine.gov/dmr/science-research/species/lobster/guide/index.html
(Accessed 3 April 2019).

State of Maine Department of Marine Resources. 2018. Maine Commercial Lobster Landings. 1–5 pp.

Statoil. 2015. Hywind Scotland Pilot Park Project: Environmental Statement.

Stebbing, P., Johnson, P., Delahunty, A., Clark, P., McCollin, T., Hale, C., and Clark, S. 2012. Reports of American lobsters, *Homarus americanus* (H. Milne Edwards, 1837), in British waters. BioInvasions Records, 1: 17–23.

Stechey, D. P. M., and Somers, K. M. 2008. Potential, realized, and actual fecundity in the crayfish *Orconectes immunis* from southwestern Ontario . Canadian Journal of Zoology, 73: 672–677.

Stelzenmüller, V., Diekmann, R., Bastardie, F., Schulze, T., Berkenhagen, J., Kloppmann, M., Krause, G., Pogoda, B., Buck, B. H. and Kraus, G. 2016. Co-location of passive gear fisheries in offshore wind farms in the German EEZ of the North Sea: A first socio-economic scoping. Journal of Environmental Management, 183: 794–805.

Stenberg, C., Støttrup, J., van Deurs, M., Berg, C., Dinesen, G., Mosegaard, H., Grome, T. and Leonhard, S. B. 2015. Long-term effects of an offshore wind farm in the North Sea on fish communities. Marine Ecology Progress Series, 528: 257–265.

Steneck, R. S. 2006. Possible Demographic Consequences of Intraspecific Shelter Competition among American Lobsters. Journal of Crustacean Biology, 26: 628–638.

Steneck, R. S., Hughes, T. P., Cinner, J. E., Adger, W. N., Arnold, S. N., Berkes, F., Boudreau, S. A., Brown, K., Folke, C., Gunderson, L., Olsson, P., Scheffer, M., Stephenson, E., Walker, B., Wilson, J. and Worm, B. 2011. Creation of a Gilded Trap by the High Economic Value of the Maine Lobster Fishery. Conservation Biology, 25: 904–912.

Steneck, R. S., and Wahle, R. A. 2013. American lobster dynamics in a brave new ocean. Canadian Journal of Fisheries and Aquatic Sciences, 70: 1612–1624.

Stockhausen, W. T., Lipcius, R. N., and Hickey, B. M. 2000. Joint effects of larval dispersal, population regulation, marine reserve design, and exploitation on production and recruitment in the caribbean spiny lobster. Bulletin of Marine Science, 66: 957–990.

Suddaby, C. 2002. Defending your livelihood with little time, no money and few rights. *In* Who Owns The Sea, pp. 44–48.

Sundelöf, A., Grimm, V., Ulmestrand, M., and Fiksen, Ø. 2015. Modelling harvesting strategies for the lobster fishery in northern Europe: the importance of protecting egg-bearing females. Population Ecology, 57: 237–251.

Szuwalski, C. S., Vert-Pre, K. A., Punt, A. E., Branch, T. A., and Hilborn, R. 2015. Examining common assumptions about recruitment: A meta-analysis of recruitment dynamics for worldwide marine fisheries. Fish and Fisheries, 16: 633–648.

Talbot, P., Thaler, C., and Wilson, P. 1984. Spawning, egg attachment and egg retention lobsters. Aquaculture, 37: 239–249.

The Crown Estate. 2017. Offshore wind energy; Project details. https://www.thecrownestate.co.uk/energy-minerals-and-infrastructure/offshore-wind-energy/our-portfolio/project-details/.

The Crown Estate. 2019. The Fishing Liaison with Offshore Wind and Wet Renewables Group. https://www.thecrownestate.co.uk/en-gb/what-we-do/on-the-seabed/our-partnerships/the-fishing-liaison-with-offshore-wind-and-wet-renewables-group/ (Accessed 4 October 2019).

The European Parliament and Council of the European Union. 2014. Directive 2014/89/EU of the European Parliament and of the Council of 23 July 2014 establishing a framework for maritime spatial planning.

Thiault, L., Kernaléguen, L., Osenberg, C.W. and Claudet, J., 2017. Progressive-Change BACIPS: a flexible approach for environmental impact assessment. *Methods in Ecology and Evolution*, 8(3), 288-296.

Thomas, R., Lello, J., Medeiros, R., Pollard, A., Seward, A., Smith, J., Vafidis, J., *et al.* 2015. Data Analysis with R Statistical Software A guidebook for Scientists. 103–112 pp.

Thomsen, F., Lüdemann, K., Kafemann, R., and Piper, W. 2008. Effects of offshore wind farm noise on marine mammals and fish. Bioacoustics, 17: 221–223.

Thorbjørnsen, S. H., Moland, E., Huserbråten, M. B. O., Knutsen, J. A., Knutsen, H., and Olsen, E. M. 2018. Replicated marine protected areas (MPAs) support movement of larger, but not more, European lobsters to neighbouring fished areas. Marine Ecology Progress Series, 595: 123–133.

Thrush, S. F., Hewitt, J. E., Cummings, V. J., and Dayton, P. K. 1995. The impact of habitat disturbance by scallop dredging on marine benthic communities: what can be predicted from the results of experiments? Marine Ecology Progress Series, 129: 141–150.

Thurstan, R. H., Brockington, S., and Roberts, C. M. 2010. The effects of 118 years of industrial fishing on UK bottom trawl fisheries. Nature Communications, 1: 1–6. Nature Publishing Group.

Titan Environmental Surveys Limited. 2013. Westermost Rough Benthic Survey. Bridgend. 1–179 pp.

Tonk, L., and Rozemeijer, M. J. C. 2019. Ecology of the brown crab (*Cancer pagurus*) : and production potential for passive fisheries in Dutch offshore wind farms. 8–37 pp.

Tully, O., Roantree, V., and Robinson, M. 2001. Maturity, fecundity and reproductive potential of the European lobster (*Homarus gammarus*) in Ireland. Journal of the Marine Biological Association of the UK, 81: 61–68.

Turner, R. a., Gray, T., Polunin, N. V. C., and Stead, S. M. 2013. Territoriality as a Driver of Fishers' Spatial Behavior in the Northumberland Lobster Fishery. Society & Natural Resources, 26: 491–505.

van Hal, R., Griffioen, A. B., and van Keeken, O. A. 2017. Changes in fish communities on a small spatial scale, an effect of increased habitat complexity by an offshore wind farm. Marine Environmental Research, 126: 26–36.

Van Marlen, B., Wiegerinck, J. A. M., van Os-Koomen, E., and van Barneveld, E. 2014. Catch comparison of flatfish pulse trawls and a tickler chain beam trawl. Fisheries Research, 151: 57–69.

Vandendriessche, S., Derweduwen, J., and Hostens, K. 2015. Equivocal effects of offshore wind farms in Belgium on soft substrate epibenthos and fish assemblages. Hydrobiologia, 756: 19–35.

Vogel, C., Kopp, D., and Méhault, S. 2017. From discard ban to exemption: How can gear technology help reduce catches of undersized *Nephrops* and hake in the Bay of Biscay trawling fleet? Journal of Environmental Management, 186: 96–107.

Waddy, S. L., and Aiken, D. E. 1990. Intermolt Insemination, an Alternative Mating Strategy for the American Lobster (*Homarus americanus*). Canadian Journal of Fisheries and Aquatatic Science, 47: 2402–2406.

Wahlberg, M., and Westerberg, H. 2005. Hearing in fish and their reactions to sounds from offshore wind farms. Marine Ecology Progress Series, 288: 295–309.

Wahle, R. A. W., Brown, C., and Hovel, K. 2013. The Geography and Body-Size Dependence of Top-Down Forcing in New England's Lobster-Groundfish Interaction. Bulletin of Marine Science, 89: 189–212.

Watson, H. V., McKeown, N. J., Coscia, I., Wootton, E., and Ironside, J. E. 2016. Population genetic structure of the European lobster (*Homarus gammarus*) in the Irish Sea and implications for the effectiveness of the first British marine protected area. Fisheries Research, 183: 287–293.

Watson, W. H., Golet, W., Scopel, D., and Jury, S. 2009. Use of ultrasonic telemetry to determine the area of bait influence and trapping area of American lobster, *Homarus americanus*, traps. New Zealand Journal of Marine and Freshwater Research, 43: 411–418.

Watson, W. H., Goldstein, J. S., Morrissey, E. M., Cole, H. A., and Pugh, T. L. 2017. Evidence of mating by sexually immature female American lobsters *Homarus americanus* (H. Milne Edwards, 1837) (Decapoda: *Nephropidae*). Journal of Crustacean Biology, 37: 2–6.

Wiber, M. G., Young, S., and Wilson, L. 2012. Impact of Aquaculture on Commercial Fisheries: Fishermen's Local Ecological Knowledge. Human Ecology, 40: 29–40.

Wickham, H. 2009. ggplot2: Elegant Graphics for Data Analysis. Springer-Verlag, New York.

Wilson, J. C., and Elliott, M. 2009. The habitat-creation potential of offshore wind farms. Wind Energy, 12: 203–212.

Wilson, J. C., Elliott, M., Cutts, N. D., Mander, L., Mendão, V., Perez-Dominguez, R., and Phelps, A. 2010. Coastal and offshore wind energy generation: Is it environmentally benign? Energies, 3: 1383–1422.

Wood, J. M. 2018. New estimates and complications in the assessment of female functional maturity for the European lobster (*Homarus gammarus*) on the Yorkshire Coast (UK). Journal of Fisheries, 6: 635–638.

Wootton, E. C., Woolmer, A. P., Vogan, C. L., Pope, E. C., Hamilton, K. M., and Rowley, A. F. 2012. Increased Disease Calls for a Cost-Benefits Review of Marine Reserves. PloS one, 7.

Wright, P. J. 2013. Methodological challenges to examining the causes of variation in stock reproductive potential. Fisheries Research, 138: 14–22.

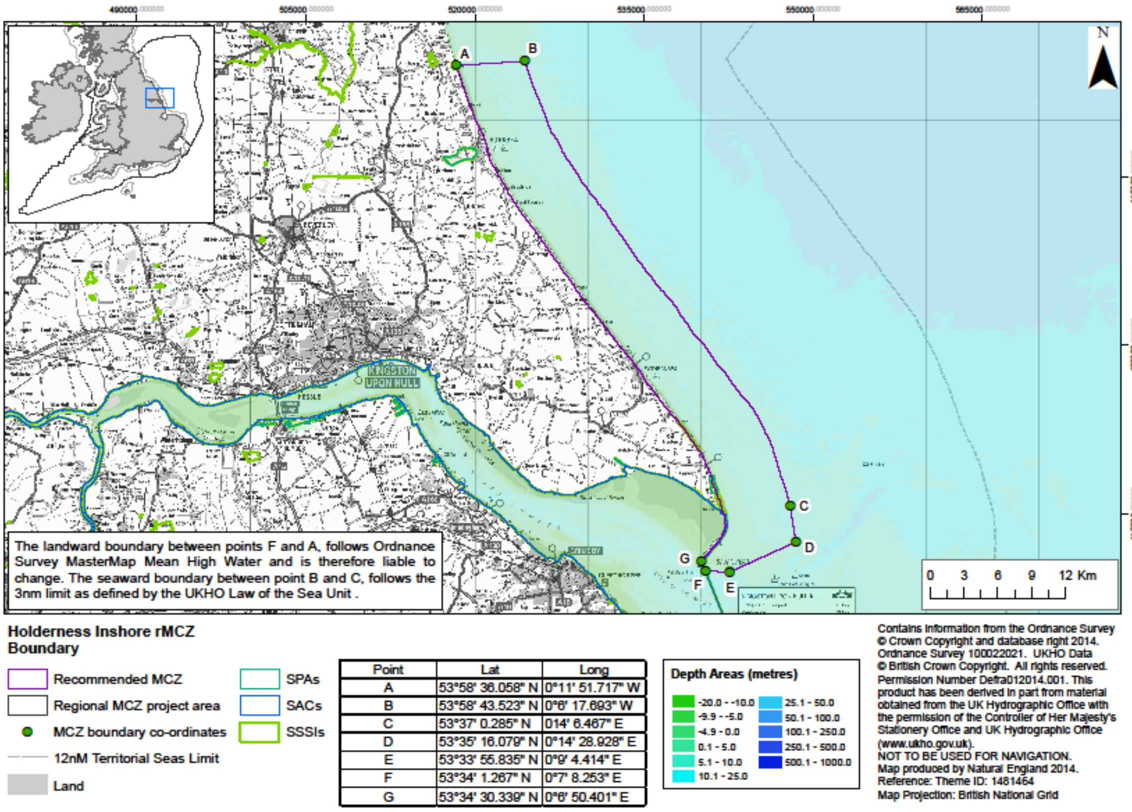
Zhang, Y., Chen, Y., and Chang, Y.-J. 2011. Estimating biological reference points using individual-based per-recruit models for the Gulf of Maine American lobster, *Homarus americanus*, fishery. Fisheries Research, 108: 385–392.

Zukowski, S., Curtis, A., and Watts, R. J. 2011. Using fisher local ecological knowledge to improve management: The Murray crayfish in Australia. Fisheries Research, 110: 120–127.

Zuur, A. F., and Ieno, E. N. 2016. A protocol for conducting and presenting results of regression-type analyses. Methods in Ecology and Evolution, 7: 636–645.

Chapter 9: Appendices

9.1 Supporting documentation for Chapter 1



Point	Lat	Long
A	53°58' 36.058" N	0°11' 51.717" W
B	53°58' 43.523" N	0°6' 17.693" W
C	53°37' 0.285" N	014' 6.467" E
D	53°35' 16.079" N	0°14' 28.928" E
E	53°33' 55.835" N	0°9' 4.414" E
F	53°34' 1.267" N	0°7' 8.253" E
G	53°34' 30.339" N	0°6' 50.401" E

Figure 9.1: Map of Holderness Inshore MCZ boundaries (DEFRA, 2019a)

9.2 Supporting statistics for Chapter 3

Table 9.1: Descriptive statistics of CPUE and LPUE for each species sampled at each site and during each year of the WMR survey.

Species	Year	Site	Unit of Effort	Mean	s.d.
Lobster	2013	Array Control	CPUE	71.39	44.67
Lobster	2013	Wind farm	CPUE	63.70	36.97
Lobster	2013	Array Control	LPUE	10.00	6.53
Lobster	2013	Wind farm	LPUE	11.13	6.98
Lobster	2015	Array Control	CPUE	107.57	29.31
Lobster	2015	Wind farm	CPUE	93.22	32.24
Lobster	2015	Array Control	LPUE	9.78	4.40
Lobster	2015	Wind farm	LPUE	22.00	15.52
Lobster	2017	Array Control	CPUE	75.20	30.23
Lobster	2017	Wind farm	CPUE	57.63	19.66
Lobster	2017	Array Control	LPUE	10.44	6.01
Lobster	2017	Wind farm	LPUE	8.75	3.51
Edible Crab	2013	Array Control	CPUE	187.35	93.42
Edible Crab	2013	Wind farm	CPUE	215.57	100.34
Edible Crab	2013	Array Control	LPUE	10.11	8.15
Edible Crab	2013	Wind farm	LPUE	10.89	8.81
Edible Crab	2015	Array Control	CPUE	91.61	55.43
Edible Crab	2015	Wind farm	CPUE	71.13	42.23
Edible Crab	2015	Array Control	LPUE	7.22	5.08
Edible Crab	2015	Wind farm	LPUE	6.30	6.28
Edible Crab	2017	Array Control	CPUE	96.75	61.03
Edible Crab	2017	Wind farm	CPUE	103.38	52.31
Edible Crab	2017	Array Control	LPUE	4.87	2.42
Edible Crab	2017	Wind farm	LPUE	6.50	3.54
Velvet Crab	2013	Array Control	CPUE	45.96	16.04
Velvet Crab	2013	Wind farm	CPUE	75.18	43.05
Velvet Crab	2013	Array Control	LPUE	30.05	13.92
Velvet Crab	2013	Wind farm	LPUE	48.92	29.72
Velvet Crab	2015	Array Control	CPUE	37.65	10.12
Velvet Crab	2015	Wind farm	CPUE	41.39	9.90
Velvet Crab	2015	Array Control	LPUE	21.78	6.98
Velvet Crab	2015	Wind farm	LPUE	24.91	6.16
Velvet Crab	2017	Array Control	CPUE	153.50	107.14
Velvet Crab	2017	Wind farm	CPUE	146.06	52.49
Velvet Crab	2017	Array Control	LPUE	91.44	68.01
Velvet Crab	2017	Wind farm	LPUE	93.12	39.04

Table 9.2: Descriptive statistics for species richness (S), individual abundance (LOGn) and Shannon Weiner diversity indices for all survey years and the wind farm and control sites.

Variable	**Site**	**Year**	**Median / *Mean***	**Range / *s.d.***
Species richness (S)	Wind farm	2013	6	3 – 10
		2015	7	4 – 9
		2017	6	3 – 8
	Control	2013	5	3 – 9
		2015	6	4 – 8
		2017	6.5	3 - 8
Individual abundance (LOGn)	Wind farm	2013	*5.51*	*0.32*
		2015	*4.96*	*0.23*
		2017	*5.25*	*0.37*
	Control	2013	*5.41*	*0.29*
		2015	*5.23*	*0.25*
		2017	*5.37*	*0.38*
Shannon Wiener diversity indices (H’)	Wind farm	2013	0.85	0.39 – 1.06
		2015	1.05	0.82 – 1.19
		2017	1.09	0.85 – 1.25
	Control	2013	0.81	0.36 – 1.10
		2015	0.97	0.82 – 1.16
		2017	1.13	0.77 – 1.33

Table 9.3: Presence and absence table of all by-catch caught during the baseline survey (2013), the first-year post build survey (2015) and third year post build survey (2017) for the WMR monitoring project.

Common Name	Scientific Name	2013	2015	2017
Shore crab	*Carcinus maenas*	✓		✓
Spiny Squat lobster	*Galathea strigosa*	✓	✓	✓
Long Clawed Squat Lobster	*Munida rugosa*	✓	✓	✓
Marbled swimming crab	*Liocarcinus marmoreus*	✓	✓	✓
Harbour Crab	*Liocarcinus depurator*	✓	✓	✓
Hermit crab	*Eupagarus bernhadus*	✓	✓	✓
Long legged spider crab	*Macropodia rostrata*	✓	✓	✓
Wrinkled Swimming Crab	*Liocarcinus corrugatus*		✓	✓
Common urchin	*Echinus esculentus*	✓	✓	✓
Common starfish	*Asteria rubens*	✓	✓	✓
Dab	*Limanda limanda*	✓	✓	✓
Sea scorpion/ Bullhead	*Myoxocephalus scorpius*	✓	✓	✓
Pouting	*Trisopterus luscus*	✓	✓	✓
Poor cod	*Trisopterus minutus*	✓	✓	✓
Cod	*Gadus morhua*	✓	✓	✓
Butter fish	*Pholis gunnelis*	✓		
Ling	*Molva molva*	✓	✓	
Whiting	*Merlangius merlangus*	✓	✓	✓
Ballan wrasse	*Labrus bergylta*	✓	✓	✓
Three bearded rockling	*Gaidropsarus vulgaris*	✓	✓	✓
Tompot blenny	*Parablennius gattorugine*	✓	✓	✓
Coley/Saithe	*Pollachius virens*		✓	✓
Pollack	*Pollachius pollachius*	✓		
Pogge/ Armoured bullhead	*Agonas cataphractus*		✓	
Red Mullet	*Mullus surmuletus*		✓	✓
Top Knot	*Zeugopterus punctatus*		✓	
Lumpsucker	*Cyclopterus lumpus*			✓

9.3 Supporting statistics for Chapter 4

Table 9.4: Results of GLMM analysing proportion of commercial catch at each size between the wind farm and control sites during each period of closure, sampled at the WMR OWF.

Scenario	Species	Model	Parameter	Estimate	Standard Error	p-value
WF and Control during **closure period**	Lobster	Linear	β_0	6.6677	0.3846	< 0.00
			β_1	-0.0812	0.0046	< 0.00
	Edible Crab	Linear	β_0	1.6966	0.3388	< 0.00
			β_1	-0.0092	0.0027	< 0.00
	Velvet Crab	Linear	β_0	4.2676	0.9275	< 0.00
			β_1	-0.0600	0.0130	< 0.00
WF and Control during **open** period	Lobster	Linear	β_0	2.0449	0.4641	< 0.00
			β_1	-0.0200	0.0058	< 0.00
	Edible Crab	Linear	β_0	-0.8784	0.3649	< 0.05
			β_1	0.0057	0.0027	< 0.05
	Velvet Crab	Linear	β_0	0.2050	0.9089	> 0.05
			β_1	-0.0063	0.0126	> 0.05

Table 9.5: Descriptive statistics of CPUE and LPUE for each species sampled at each site and during closure period of the WMR OWF survey in 2015.

WF Status	Site	Effort	Species	Mean	SD
Closed	WF	CPUE	Lobster	113.08	29.31
Closed	Control	CPUE	Lobster	107.08	35.44
Closed	WF	LPUE	Lobster	36.83	10.43
Closed	Control	LPUE	Lobster	12.08	4.23
Open	WF	CPUE	Lobster	71.73	18.59
Open	Control	CPUE	Lobster	107.55	22.98
Open	WF	LPUE	Lobster	8.73	6.25
Open	Control	LPUE	Lobster	8.27	4.47
Closed	WF	CPUE	Edible Crab	64.0	37.97
Closed	Control	CPUE	Edible Crab	113.54	61.68
Closed	WF	LPUE	Edible Crab	6.07	6.47
Closed	Control	LPUE	Edible Crab	7.69	4.35
Open	WF	CPUE	Edible Crab	80.40	44.68
Open	Control	CPUE	Edible Crab	62.40	21.40
Open	WF	LPUE	Edible Crab	8.90	6.76
Open	Control	LPUE	Edible Crab	7.70	7.42
Closed	WF	CPUE	Velvet Swimming Crab	38.15	9.41
Closed	Control	CPUE	Velvet Swimming Crab	39.08	10.69
Closed	WF	LPUE	Velvet Swimming Crab	30.15	4.49
Closed	Control	LPUE	Velvet Swimming Crab	29.23	9.98
Open	WF	CPUE	Velvet Swimming Crab	37.00	9.82
Open	Control	CPUE	Velvet Swimming Crab	47.20	9.53
Open	WF	LPUE	Velvet Swimming Crab	36.80	8.52
Open	Control	LPUE	Velvet Swimming Crab	29.40	8.53

Table 9.6: Presence and absence table of all by-catch caught during the 2015 survey. By-catch data has been separated to represent both sites and the open and closed period.

Common Name	Scientific Name	2015	Control Closed	WF Closed	Control Open	WF Open
Shore crab	*Carcinus maenas*					
Spiny Squat lobster	*Galathea strigosa*	✓		✓	✓	✓
Long Clawed Squat Lobster	*Munida rugosa*	✓		✓	✓	✓
Marbled swimming crab	*Liocarcinus marmoreus*	✓	✓	✓	✓	✓
Harbour Crab	*Liocarcinus depurator*	✓	✓	✓	✓	✓
Hermit crab	*Eupagarus bernhadus*	✓	✓	✓	✓	✓
Long legged spider crab	*Macropodia rostrata*	✓	✓	✓	✓	✓
Wrinkled Swimming Crab	*Liocarcinus corrugatus*	✓	✓	✓	✓	✓
Common urchin	*Echinus esculentus*	✓	✓		✓	✓
Common starfish	*Asteria rubens*	✓	✓			
Dab	*Limanda limanda*	✓	✓		✓	✓
Sea scorpion/ Bullhead	*Myoxocephalus scorpius*	✓	✓	✓		✓
Pouting	*Trisopterus luscus*	✓	✓	✓	✓	
Poor cod	*Trisopterus minutus*	✓	✓	✓	✓	✓
Cod	*Gadus morhua*	✓	✓	✓	✓	✓
Butter fish	*Pholis gunnelis*					
Ling	*Molva molva*	✓			✓	✓
Whiting	*Merlangius merlangus*	✓		✓	✓	✓
Ballan wrasse	*Labrus bergylta*	✓	✓	✓	✓	✓
Three bearded rockling	*Gaidropsarus vulgaris*	✓	✓	✓		
Tompot blenny	*Parablennius gattorugine*	✓	✓	✓	✓	✓
Coley/Saithe	*Pollachius virens*	✓	✓	✓	✓	✓
Pollack	*Pollachius pollachius*					
Pogge/ Armoured bullhead	*Agonas cataphractus*	✓				✓
Red Mullet	*Mullus surmuletus*	✓	✓	✓	✓	✓
Top Knot	*Zeugopterus punctatus*	✓		✓		
Lumpsucker	*Cyclopterus lumpus*					

9.4 R Code used for GLMM

9.4.1 Auxiliary function for the model building

```
get.data.glmm=function(dat,spec,geartest){
  # Function that extracs data for glmer

test=dat[(dat$SPEC==spec)&(dat$GEAR==geartest)&(dat$COMPARTMENT=="TEST"),c("HAU
L","LENGTH","COUNT","SUBSRATIO")]

ctrl=dat[(dat$SPEC==spec)&(dat$GEAR==geartest)&(dat$COMPARTMENT=="CONTROL"),c("
HAUL","LENGTH","COUNT","SUBSRATIO")]
  loc.dat=merge(test,ctrl,by=c("HAUL","LENGTH"),all=T)[,c(2,3,5,1,4,6)]
  q=loc.dat[,5:6]
  q[is.na(q)]=1
  loc.dat[is.na(loc.dat)]=0
  loc.dat=cbind(loc.dat[,1:4],q)
  names(loc.dat)=c("LENGTH", "TEST", "CTRL", "HAUL","q.TEST","q.CTRL")
  loc.dat
}

ilogit=function(eta){1/(1+exp(-eta))}

get.sel.and.conf.band.catch=function(parm,varm,l.min,l.max,n=200){
  lgt<-seq(l.min,l.max,length=n)
  X=sapply(1:length(parm),function(i){lgt^(i-1)})
  reg.line=X%*%parm
  se=apply(X,1,function(x,varm){sqrt(t(x)%*%varm%*%x)},varm)
  cbind(lengtht=lgt,min.L50=reg.line-2*se,mean.L50=reg.line,max.L50=reg.line+2*se)
}
```

9.4.2 Function to generate the GLMM plot

```
plot.fit=function(glmm.fit,limx,labx="",laby="",Spec.Text="",header=""){
  varm=as.matrix(vcov(glmm.fit))
  coeff=fixef(glmm.fit)
  eta.matrix=get.sel.and.conf.band.catch(coeff,varm,min(limx),max(limx))
  xxx=c(eta.matrix[,1],rev(eta.matrix[,1]))
  yyy=c(ilogit(eta.matrix[,2]),rev(ilogit(eta.matrix[,4])))
  plot(eta.matrix[,1],ilogit(eta.matrix[,3])
       ,type="l",xlab=labx, ylab=laby, ylim=c(0,1),
xlim=limx,cex.lab=1.5,cex.axis=1.5,col="black",lwd=2, las=1)
  polygon(xxx,yyy,col="grey",density=NULL,border="black")
  lines(eta.matrix[,1],ilogit(eta.matrix[,3]),col=1,lwd=2)
  abline(h=0.5,lty=1)
  if (Spec.Text!="")
    text(sum(limx)/2,0.9,Spec.Text,cex=1.7)
  if (header!="")
    mtext(header,cex=1.3)
  #axis(3)
}
```

9.4.3 Model building and plotting

```
fit.lob.lin=glmer(cbind(TEST,CONTROL)~1+Length+Sex+Berried+Grade+(1|Haul),
family=binomial,data=dat.wide,sub=Species=="Lobster")

summary(fit.lob.lin)
```

```
plot.fit(fit.lob.lin,c(40,130),labx="Carapace Length",laby="Proportion of animals retained",
Spec.Text=="",header="Lobster")
```